IAEA-BIOMASS-1

Modelling the migration and accumulation of radionuclides in forest ecosystems

Report of the Forest Working Group of the Biosphere Modelling and Assessment (BIOMASS) Programme, Theme 3

August 2002

INTERNATIONAL ATOMIC ENERGY AGENCY

The originating Section of this publication in the IAEA was:

Waste Safety Section
International Atomic Energy Agency
Wagramer Strasse 5
P.O. Box 100
A-1400 Vienna, Austria

MODELLING THE MIGRATION AND ACCUMULATION OF
RADIONUCLIDES IN FOREST ECOSYSTEMS
IAEA, VIENNA, 2002
ISBN 92–0–114902–6

Printed by the IAEA in Austria
August 2002

FOREWORD

The IAEA Programme on *BIO*sphere *M*odelling and *ASS*essment (BIOMASS) was launched in Vienna in October 1996. The programme was concerned with developing and improving capabilities to predict the transfer of radionuclides in the environment. The programme had three themes:

Theme 1: Radioactive Waste Disposal. The objective was to develop the concept of a standard or reference biosphere for application to the assessment of the long-term safety of repositories for radioactive waste. Under the general heading of "Reference Biospheres", six Task Groups were established:

Task Group 1: Principles for the Definition of Critical and Other Exposure Groups.

Task Group 2: Principles for the Application of Data to Assessment Models.

Task Group 3: Consideration of Alternative Assessment Contexts.

Task Group 4: Biosphere System Identification and Justification.

Task Group 5: Biosphere System Descriptions.

Task Group 6: Model Development.

Theme 2: Environmental Releases. BIOMASS provided an international forum for activities aimed at increasing the confidence in methods and models for the assessment of radiation exposure related to environmental releases. Two Working Groups addressed issues concerned with the reconstruction of radiation doses received by people from past releases of radionuclides to the environment and the evaluation of the efficacy of remedial measures.

Theme 3: Biosphere Processes. The aim of this Theme was to improve capabilities for modelling the transfer of radionuclides in particular parts of the biosphere identified as being of potential radiological significance and where there were gaps in modelling approaches. This topic was explored using a range of methods including reviews of the literature, model inter-comparison exercises and, where possible, model testing against independent sources of data. Three Working Groups were established to examine the modelling of: (1) long-term tritium dispersion in the environment; (2) radionuclide uptake by fruits; and (3) radionuclide migration and accumulation in forest ecosystems.

This report describes results of the studies undertaken by the Forest Working Group under Theme 3. The IAEA Scientific Secretary for this publication was F. Gera of the Division of Radiation and Waste Safety. The IAEA wishes to acknowledge the contribution of the Forest Working Group Leader, George Shaw of the United Kingdom, to the preparation of this report. Additional financial support was provided to this group by Statens Stralskyddinstitut (SSI), Sweden; Agence Nationale pour la Gestion des Déchets Radioactifs (ANDRA), France; British Nuclear Fuels plc (BNFL), United Kingdom; Centro de Investigaciones Energéticas Medioambientales y Tecnológicas (CIEMAT) and Empresa Nacional de Residuos Radiactivos SA (ENRESA), Spain; Institut de Protection et de Sûreté Nucléaire (IPSN), France; Nationale Genossenschaft für die Lagerung radioaktiver Abfälle (NAGRA), Switzerland; Japan Nuclear Cycle Development Institute (JNC), Japan and United Kingdom Nirex Limited (Nirex), United Kingdom.

EDITORIAL NOTE

CONTENTS

SUMMARY

This publication describes the work carried out during a two year period from 1998 to 2000 by the IAEA's BIOMASS Forest Working Group (Forest WG). The primary objective of the Forest WG's activities was to bring together modellers working in the field of radionuclide behaviour in forest ecosystems and to facilitate their interaction in the following areas:

- Review and exchange of existing information and peer review.
- Development of a list of features, events and processes (FEPs) for forests using an interaction matrix approach.
- Inter-comparison of existing models to identify and investigate significant areas of uncertainty and differences in modelling approach.
- Testing and validation of existing or new models against independent data sets, where available.
- Recommendation for possible future directions in the modelling of radionuclide behaviour in forests.

The work of the Forest WG began with a review of the biogeochemical cycling of radiocaesium in forest ecosystems, based on the best available knowledge obtained both before and after the Chernobyl accident of 1986. It was concluded from this review that there is still a need for a standardization of current approaches to data acquisition in a format which can be used to facilitate the comparison of radionuclide cycling in forests at different geographical locations and, preferably, on a whole-ecosystem basis.

The application of an interaction matrix method to forest model design was investigated as a means of both improving the design of models based on best available knowledge and achieving a greater degree of consensus between modelers on the objectivity and fitness for purpose of forest models. The definitions and application of transfer factors to forests was also reviewed in some detail and a new method of determining radiocaesium uptake within the woody tissues of trees (the wood immobilization potential, or WIP) was proposed.

The major part of the Forest WG's work was devoted to conducting three model inter-comparison studies based on three different scenarios. It was considered that an early emphasis on such practical modelling exercises was desirable, so an inter-comparison of the predictions of ten models for a hypothetical Chernobyl-type scenario was conducted within the first six months of the Forest WG's activities. The results indicated a generally high level of consistency between model predictions for 'structural components' of the forest such as soils and trees. Predictions for biological endpoints such as edible mushrooms, however, were more variable. A detailed statistical analysis of these results was carried out.

A second model inter-comparison exercise was carried out using previously unseen data obtained from the Zhitomir region of Ukraine, which became contaminated with ^{137}Cs in 1986. Blind predictions were made by nine modellers and the results revealed a similarly high level of consistency between model predictions as in the first scenario. Many of the model predictions proved rather accurate, although the available data only covered a period of some 5 to 12 years after initial contamination when the early dynamics of ^{137}Cs within the forest ecosystem are likely to have been superseded by slower, long-term rates of redistribution. Particularly accurate and consistent predictions were made for the tree-related compartments

and for certain soil compartments. As in the case of the first scenario, a detailed statistical analysis of these results was carried out.

The third and final modelling exercise was again hypothetical and involved comparison of model outputs against each other. Whereas the first two scenarios had involved Chernobyl-type scenarios, however, this scenario involved a shallow landfill type repository as a source term. Even though none of the Forest WG models had been constructed with this type of scenario in mind, seven models were used to provide predictions in this exercise, although some degree of model re-coding was required by at least some of these models. The degree of agreement between predictions for tree components and for biological endpoints was striking, although the reasons for this are not immediately evident. A key question arising from the results obtained is whether physical or biological transport of ^{137}Cs through the soil is likely to dominate when the source is below the ground surface and when deep-rooted plants, such as trees, are allowed to access such subterranean sources.

The conclusions of the Forest WG's activities are finally summarized and recommendations for future experimental and modelling studies within the broad field of forest radioecology are made. These include the issue of time-dependency in forest processes (including tree growth), high versus low deposition scenarios, process-orientated models, integration of forest radioecology with other ecological models, and the question of whether it is possible to construct a 'generic' model of radionuclide behaviour in forests.

Appended to this report are summary descriptions of the individual models which participated in the Forest WG modelling studies, as well as detailed descriptions of each of the modelling scenarios addressed by the Forest WG.

1. INTRODUCTION

1.1. BACKGROUND

Forests are extensive natural resources that provide economic, nutritional, recreational and social benefits to people in many countries. Following the Chernobyl accident, radioactive contamination of forests occurred in Ukraine, Belarus and Russia, and also in many countries beyond the borders of the former Soviet Union, notably Finland and Sweden. The degree of contamination with ^{137}Cs in these countries ranged from >10 MBq m^{-2} in some locations to between 10 and 50 kBq m^{-2}, the latter range being typical of ^{137}Cs deposition to several countries of western Europe. In each of these countries, not only do forests provide an economic resource of major importance, but they are at the heart of many social and cultural activities which, in some cases, have been curtailed by the deposition of ^{137}Cs in 1986. Despite such problems, when compared with the research efforts committed to understanding the radiological impact of ^{137}Cs contamination of agriculture, contaminated forest ecosystems have remained on the periphery of interest of many national radiological protection organisations.

In the fourteen years since the Chernobyl accident it has become apparent in countries across Europe and the former Soviet Union that natural decontamination of contaminated forests is proceeding extremely slowly. Since net export of ^{137}Cs from forest ecosystems has been determined to be less than 1% per year [Tikhomirov et al., 1993, Nylen, 1996], it is likely that, without artificial intervention, it is the physical decay rate of ^{137}Cs that will determine the duration over which forests continue to be affected by the Chernobyl legacy. Despite the fact that the absolute natural losses of ^{137}Cs from the forest are minimal, recycling of ^{137}Cs within the forest is a dynamic process in which reciprocal transfers occur on a seasonal, or longer-term, basis between biotic and abiotic components of the ecosystem. In order to facilitate long-term management of forests which may, potentially, represent a radiological hazard to both human and non-human populations, a reliable understanding of these exchange processes is required. Primary information on such processes is obtained from experiments and field measurements, but for highly diverse systems such as forests it is inevitable that this information can be better assimilated in a holistic sense through the application of mathematical models.

Radioecological models can be used to simulate system responses to radionuclide inputs and system manipulations of various kinds and, thereby, to assist in the assessment of different contamination scenarios and post-contamination management options. For any reliability to be placed on system simulations, however, it is essential that models are correctly scrutinised in appropriate validation studies. Validation exercises involving more than one model and more than one group of model users are likely to yield the most satisfactory results since this approach should ensure that an individual model is tested in scenarios which are not restricted to the original situations or cases for which that model was originally developed. The activities of the Forest Working Group (Forest WG) within the IAEA's BIOMASS programme were designed to provide an international platform on which forest radioecology models could be tested.

The Forest WG activities within the IAEA BIOMASS programme have their origins in the preceding VAMP (Validation of Environmental Model Predictions) programme, which ran from 1988 to 1993. One of the themes within the Terrestrial Working Group of VAMP was 'Food Chain Transfer in Natural and Semi-Natural Ecosystems'. Forests belong to this group of ecosystems and a small Forest Group began to address the problem of appropriate model design and data sources as part of the working group's activities. Unfortunately, at that time, there were no suitable radioecological models available for forests which could be put forward for a validation study. However, by the time the BIOMASS programme began this situation had changed radically. It was evident in 1997/98 that several research groups across Europe and North America had developed models which could be used to simulate radiocaesium behaviour in forest ecosystems (e.g. in EU Projects such as Seminat and Landscape [Belli 2000; Brechignac et al. 2000]).

1.2. OBJECTIVES

The main purpose of the BIOMASS Forest WG was to provide a forum for inter-comparison and validation of models designed to simulate the ecosystem behaviour of radionuclides, especially ^{137}Cs, within forest ecosystems of the temperate and boreal latitudes.

Five specific objectives were drawn up for the Forest WG, as follows:

(1) To bring together modellers in the field of radionuclide transfer within forests to facilitate exchange of information and peer review.

(2) To undertake model inter-comparisons to identify and investigate significant areas of uncertainty and differences in approach.

(3) Where possible and practicable, to undertake testing and validation of existing or new models against independent data sets.

(4) To take account of the wider implications of the specific results of the Forest WG by developing a list of features, events and processes (FEPs) for forests.

(5) Based on the above, to make recommendations for the future direction of the modelling of radionuclides in forests.

1.3. SCOPE

It is important to recognise that the model testing activities listed in Section 1.2 are closely associated with model building and data gathering. The activities of the Forest WG have also provided an opportunity to consolidate existing and new information on such key issues as biogeochemical cycling within forests and the use and abuse of transfer parameters. Over and above these activities, however, the Forest WG has attempted to bring together forest modellers and experimentalists from 12 countries to share their models, their data and their opinions on the status of forest radioecological modelling at present and where it should be heading in the future.

1.4. STRUCTURE

This report provides a summary of the activities of the Forest WG, based on the above objectives, between 1998 and 2000. Sections 2, 3 and 4 are devoted to reviews of the biogeochemical cycling of radiocaesium in forest ecosystems, the application of the interaction matrix method to forest model design, the definition and application of transfer factors to forests and a new method of determining radiocaesium uptake within the woody tissues of trees. Sections 5, 6 and 7 describe the three model inter-comparison exercises (Scenarios 1, 2 and 3, respectively) carried out with the participation of 11 modelling groups. Finally, Sections 8 and 9 summarise the main conclusions from the activities of the Forest WG and make recommendations for future experimental and modelling studies within the broad field of forest radioecology.

Appended to the report are summary descriptions of the individual models that participated in the Forest WG modelling studies, as well as detailed descriptions of each of the modelling scenarios addressed by the Forest WG.

2. RADIOCAESIUM CYCLING IN FOREST ECOSYSTEMS

2.1. BIOGEOCHEMICAL CYCLING OF RADIOCAESIUM IN FORESTS

The study of elemental cycling within environmental systems allows us to determine the dominant processes controlling the transport of elements in the system and is usually a prerequisite to modelling the system. Normally, a conceptual biogeochemical cycle for a forest ecosystem is represented as the combination of two interconnected cycles, namely the geochemical and biological cycles. The geochemical cycle is an open cycle comprising two main components – the input and output of an element into or out of the forest system. The geochemical cycle is, in fact, intimately connected to the biological cycle which comprises the fluxes of elements mobilised during biomass growth and degradation. The schematic figure below (Figure 1) illustrates a generally accepted forest biogeochemical cycle, applicable to radiocaesium, in terms of the major compartments and fluxes.

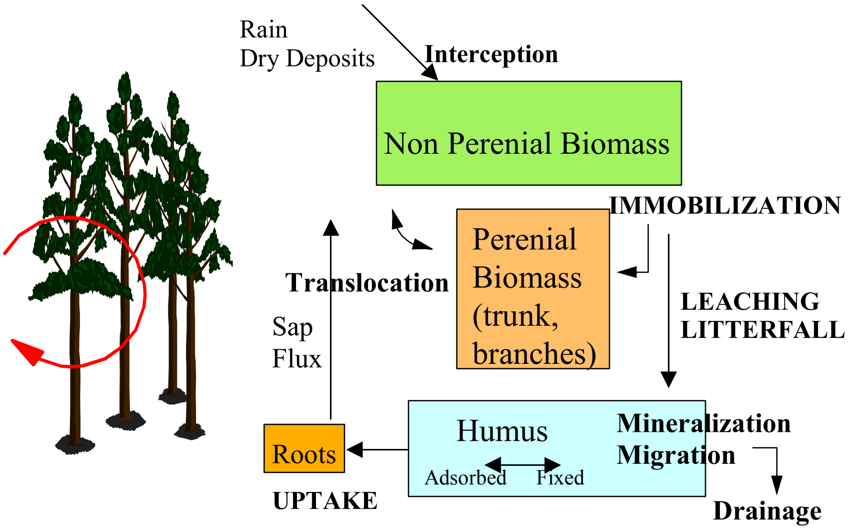


FIG. 1. A general scheme of the biogeochemical cycling of radiocaesium within a forest ecosystem.

Most of the recent modelling developments in forest radioecology deal with the cycling of ^{137}Cs deposited in 1986 as a result of the Chernobyl accident. Recent reviews and analyses of information available on processes and models of radiocaesium cycling in forests have been provided through peer-reviewed publications [Thiry and Myttenaere 1993; Myttenaere et al. 1993; Nimis 1996; Mamikhin et al. 1997; Avila et al. 1998; Ipatyev et al. 1999 and Thiry et al. 2000], within conference proceedings [Linkov and Schell, 1999; Riesen et al. 1999; and Delvaux et al. 1999] or in PhD theses [Linkov 1995; Thiry 1997 and Ravila 1998].

Following deposition of ^{137}Cs from the Chernobyl plume, the primary source of tree contamination (60–90%) was direct interception of aerosol-associated radiocaesium by the canopy, followed by further translocation from foliar surfaces to structural components of the tree. Further changes in tree contamination after the initial fallout was due to two main processes. The first of these was a dominant and relatively rapid self-decontamination process of the tree canopy, effected by precipitation wash-off (throughfall) and litterfall, and this was followed by root uptake which has prevailed over the longer term as radiocaesium has migrated into the soil profile. Just as in the case of its nutrient analogue, potassium, the rate of radiocaesium cycling within forests is relatively rapid and quasi-equilibrium of its distribution is probably reached a few years after atmospheric fallout. The upper, organic-rich, soil layers act as a long term sink but also as a general source of radiocaesium for contamination of forest vegetation, although individual plant taxa differ greatly in their ability to accumulate radiocaesium from this organic soil. Output from the system via the drainage water is generally limited as a result of radiocaesium fixation on micaceous clay minerals [Nylen 1996]. An important role of the vegetation in the recycling of radiocaesium in the forest is the partial and transient storage of radiocaesium, particularly in perennial woody components such as tree trunks and branches that can have a large biomass. The major portion of radiocaesium accumulated by vegetation from the soil, however, is recycled annually through leaching and litterfall, resulting in the long-lasting biological availability of radiocaesium in surface soil. Internal translocation of radiocaesium within vegetation also occurs, but involves generally low radiocaesium activities compared with exchange (uptake/return) between the soil and the forest vegetation.

2.2. TOWARDS A GENERAL, CONCEPTUAL MODEL OF RADIOCAESIUM CYCLING WITHIN FORESTS

As implied by the preceding section, biogeochemical cycling integrates most of the driving functions of the ecosystem and its study thus involves the ecosystem as a whole. Ecosystem functioning involves, *inter alia*, a continual exchange of elements between various biotic and abiotic components. These exchanges also apply to trace contaminants like radiocaesium, the rate of recycling of which varies with factors such as climate, soil type, vegetation species and with the stage of tree development.

Different interpretations of the spatial and temporal complexities of forest ecosystems can result in different conceptual models of biogeochemical functioning of a forest. A simplified model was proposed by Ulrich [1973], developed by Cole and Rapp [1981] and refined by Ranger and Bonneau [1984]. This model stratifies the ecosystem into compartments delineated by the specificity and homogeneity of their respective fate. The main components are diverse compartments of the vegetation, the soil organic and mineral horizons and their soil solution, and certain key fluxes which are directly measurable (i.e. atmospheric inputs, leaching and water drainage, litterfall). A mass inventory of elements can be defined for each compartment and fluxes are deduced from input–output measurements. Certain fluxes which are not directly measurable, such as root uptake, can be calculated. This approach allows us to quantify the dominant elemental fluxes within the ecosystem and to establish an overall elemental balance. Furthermore, when this approach is applied to a suitable forest chronosequence [Cole and Van Miegroet 1989], the effect of forest age and stage of development on elemental fluxes can be estimated.

During the last decade, numerous studies of radiocaesium transfer have been conducted at a wide range of forest sites located across western Europe and the countries of the CIS. In such studies, the extent of radiocaesium exchange between forest compartments such as soil and vegetation is most often characterised through the determination of a transfer coefficient or factor (TF) even though the use of the TF to quantify the dynamics of radiocaesium exchanges in forest ecosystems presents obvious limitations. Even at those sites where the dynamics of radiocaesium cycling have been studied, the bulk of research has often been limited to single biogeochemical pathways rather than examining the system *in toto*. In recently published reviews, therefore, reasonably complete data for radiocaesium cycling within forests have not been widely available. Thus, there is still a need for a standardization of current approaches to data acquisition in a format which can be used to facilitate the comparison of radiocaesium cycling at different geographical locations and, preferably, on a whole-ecosystem basis. An alternative way to improve the development of conceptual models is the application of a systematic method of identifying dominant features, events and processes (FEPs) using an ‘interaction matrix’ approach [Avila and Moberg 1999]. The reliability of simulation models depends largely on our knowledge of individual processes represented within the model and the way in which these interact. Objectivity in identifying and characterising these processes can be increased by involving a number of experts in creating and coding the matrices.

The advantages of the ‘interaction matrix’ approach for designing and testing mathematical simulation models, and the adequacy of transfer factors to assess radiocaesium redistribution in specific biological components of forest ecosystems, are discussed below in Section 2.3 and Section 3, respectively.

2.3. A SYSTEMATIC APPROACH TO THE MIGRATION OF ^{137}Cs IN FOREST ECOSYSTEMS USING INTERACTION MATRICES

As described above, the migration of radionuclides in a forest ecosystem is the result of multiple biotic and abiotic interactions between many components. When developing conceptual and mathematical models of such complex systems, there is always a risk that important components and/or interactions will be omitted or underestimated. This risk can be reduced if a systematic approach to model development is applied, for example by using interaction matrices [Avila and Moberg 1999]. This method (the matrix method) was used in the BIOMASS Forest WG to develop a general conceptual model of radiocaesium migration in the forest, which was afterwards used as a reference for identifying differences and similarities between the compared models.

The essence of the matrix method is to study processes occurring in a system using an interaction matrix. In such a matrix the components of the system in question are elements in the leading diagonal (top left to bottom right) and the interactions between these are the off-diagonal elements. The number of diagonal elements will be a measure of the resolution of the matrix (i.e. the degree of complexity, or simplicity, of the interpretation of the system by the modeller). The larger the number of leading diagonal terms, the higher the number of possible interaction terms and the higher the resolution. For a matrix with N diagonal terms, there are N (N-1) interaction terms. A key question determining the complexity of the model developed as a result of the matrix development is the selection of the diagonal elements and the optimal resolution based on the modeller’s knowledge about the system. There is no universal objective method for this and, thus, the selection of diagonal elements strongly depends on the expert judgement of the person building the matrix. One way to increase the objectivity of this process is to involve several experts, for instance a multidisciplinary group such as the Forest WG. To make the matrix more useful for studies of cause–effect relationships, pathway analysis, etc., the diagonal elements should be selected in such a way that as many binary interactions as possible are placed in off-diagonal elements. The resulting matrix can be checked for completeness by assuming that each binary interaction is, in principle, possible and by a comparison with information on these interactions to be found in the literature.

The matrix shown in Figure 2 was developed by the BIOMASS Forest WG. This matrix can be regarded as the general consensus within the group of the principal transfer processes that are relevant and necessary to describe the migration of ^{137}Cs in a forest ecosystem. It also shows a conceptualisation that was considered suitable to represent the interactions prevailing in the system. Hence, the matrix in Figure 2 is also a conceptual representation of a compartmental model of radiocaesium cycling in a forest ecosystem, in which the diagonal elements represent the storage compartments and the off-diagonal elements represent the transfer pathways between these compartments.

The derivation of a mathematical model from an original conceptual model is not a straightforward task and different modellers will have their own approaches to this problem. When the modelled system is complicated it is difficult to foresee which level of aggregation is optimal and which interactions or pathways should be included. A rather common approach is to start with a very detailed model (a ‘research’ model), which is afterwards simplified by sensitivity analyses and screening procedures. However, this method can not always be applied due to a lack of knowledge of parameter values for the many exchanges and interactions which a research model usually involves. Such models also require large calculation efforts.

Atmos phere	intercept. rainfall snowfall	intercept. rainfall snowfall			intercept. rainfall, snowfall			intercept. rainfall snowfall	intercept. rainfall snowfall	intercept. inhalation
Transpir.	Tree leaves	weathering	translocation	translocation	leaf fall, weathering			weather. intercept.	weather. intercept.	ingestion
		External bark	translocation		weather. intercept.			weather. intercept.	weather. intercept.	ingestion
	translocation	translocation	Living wood	translocation		fertilisation	fertilisation	mycorrhizae transfer		ingestion
			translocation	Dead wood						
Resuspension		rain splash	root uptake		Litter	Decomp. Percolation soil biota		uptake	rain splash, root uptake	ingestion
			root uptake			Soil organic	percolat. diffusion Advect., soil biota	uptake	root uptake	
			root uptake			diffusion, capillary rise, soil biota	Soil Mineral	uptake	root uptake	
			root upt. (mycorrhizae)		fertilisation	fertilisation	fertilisation	Fungi	Root upt. (mycorrhizae)	ingestion
transpir.					leaf fall, weather. intercept.	fertilisation	fertilisation	mycorrhizae transfer	Under storey	ingestion
					fertilisation					Game

FIG. 2. An interaction matrix which describes the migration of ^{137}Cs in a forest ecosystem. The diagonal elements are components of the system (i.e. model compartments) and the off-diagonal elements are the interactions between them (transfer processes between compartments). In order to identify the transfer processes the matrix should be read clockwise.

An alternative method is to start the process by building a simple compartment model, which includes only the major pathways. These can be identified after ranking the pathways with the help of an interaction matrix and an initial version of the model would include only those pathways with the highest rank. Thereafter, other pathways can be systematically added to this model according to their rank. After each addition the effect on a predefined optimisation function is evaluated. The uncertainty of the estimations, or the differences between the estimations and a set of experimental data, are examples of possible optimisation functions. The addition of pathways ends when the desired level of the optimisation is reached. So, the matrix in Figure 2 can be used by modellers as a starting point for the derivation of models with more specific endpoints. Other practical uses of this matrix are discussed by Avila and Moberg [1999].

2.4. COMPARISON OF FOREST WORKING GROUP MODELS WITH MATRIX

The models which participated in the three inter-comparison studies undertaken by the BIOMASS Forest WG are listed in Table 1. A brief description of each model can be found in Annex I. The forest ecosystem compartments (and intercompartmental transfers) depicted in the matrix in Figure 2 are represented to a varying degree in each of these models. Some of the 11 compartments and 75 transfers depicted in Figure 2 are represented explicitly in the models, but others are only represented implicitly. Explicit/implicit representation of compartments and transfers in each model as a percentage of the total is shown graphically in Figures 3 and 4.

The degree of representation by each of the participating models of compartments and transfer processes identified in the matrix is of interest since, as stated in Section 2.3, the matrix represents a consensus view of what an idealised forest model should represent. Deviations from this idealised model could lead to enlargement of Type B uncertainty in individual models, i.e. those uncertainties due to incorrect representation of compartments or processes within a model [Hoffman and Hofer, 1988]. Explicit representation of the 11 compartments identified in the matrix varied from 90% to 45% in the models included in the Forest WG inter-comparison exercises. This discrepancy, which indicates the variability in the level of complexity of each of the models, can be explained by the fact that the models identified in Table 1 were either conceived as general forest radioecology models, which attempt to model the behaviour of radiocaesium in the forest system as a whole, or as more focussed models intended to address specific transport pathways, such as soil migration and uptake of radiocaesium by understorey vegetation. Hence, models such as FORESTLAND, FOA and FORM, each designed to simulate radiocaesium behaviour in the forest system as a whole, represent explicitly 90% of the 11 possible compartments. In the case of the FOA model, the remaining 10% of compartments is represented implicitly. In contrast, FORSUN represents explicitly only 45% of the 11 compartments identified in the matrix. This model was more narrowly focussed on the problem of soil–plant transfer of radiocaesium to understorey vegetation in the forest ecosystem.

While it is perhaps straightforward for modellers to agree on the notional compartments of the forest ecosystem which should be represented within their idealised model, it is far less certain which of the processes effecting transfer between these compartments should be represented. Indeed, it is often the case that the operation and significance of specific transfer processes are in doubt, due either to a lack of fundamental knowledge concerning the process or a lack of quantitative data on the rates of individual processes. A very good example is the role of mycorrhiza in the transfer of radiocaesium from forest soils to trees and understorey vegetation.

TABLE 1. MODELLERS AND MODELS PARTICIPATING IN THE BIOMASS FOREST WG INTER-COMPARISON STUDIES

Modeller(s)	Model	Institute
R. Avila and L. Moberg	FORESTLAND	SSI, Stockholm, Sweden
G. Shaw	RIFE	Imperial College, UK
S. Fesenko and S. Spiridonov	FORESTLAND	RIARAE, Russia
R. Bergman	FOA	NDRE, Umea, Sweden
P. Calmon	RODOS	IPSN, France
A. Dvornik and T. Zhuchenko	FORESTLIFE	BFI, Gomel, Belorus
M. Frissel	FORM	Consultant (IAEA)
I. Linkov	FORESTPATH	Harvard University, USA
S. Mamikhin	ECORAD-C	MSU, Moscow, Russia
A. Rantavaara	FINNFOOD	STUK, Helsinki, Finland
A. Konoplev and A Bulgakov	FORSUN	Typhoon, Obninsk, Russia
A. Konoplev and A Bulgakov	FORWASTE	Typhoon, Obninsk, Russia
A. Rantavaara and J Wendt	S-RODOS[1]	STUK, Helsinki, Finland
M. Scimone	LOGNAT	Trieste, Italy

The general uncertainty surrounding the significance of many of the possible transfer processes identified in Figure 2 is reflected in the rather low degree of representation of the 75 processes shown in the matrix in the models themselves. Explicit representation of these processes ranged from 50% in the FOA model to only 10% in the RIFE model. Figure 4 also indicates that in some models (RIFE and ECORAD-C) the implicit representation of transfer processes accounted for 50% or more of the total number of transfers included in the model. This indicates that lumping together of transfer processes into single effective transfer rates or transfer coefficients is a common and perhaps inevitable approach to modelling transfer of radionuclides in complex ecosystems such as forests for which there is still incomplete information on transfer rates and processes (see Section 3).

This comparison of the models used within the Forest WG with the matrix shown in Figure 2 illustrates that, while it is certainly possible to obtain agreement within a diverse group of modellers on which components and transfer processes should be represented in an idealised forest radioecology model, the working models that individual modellers actually build are inevitably designed and modified to suit the individual needs of the modeller. The ‘ideal’ forest radioecology model has yet to be constructed.

---

[1] S-RODOS is a dynamic submodel for tree and understorey calculations in the RODOS model.

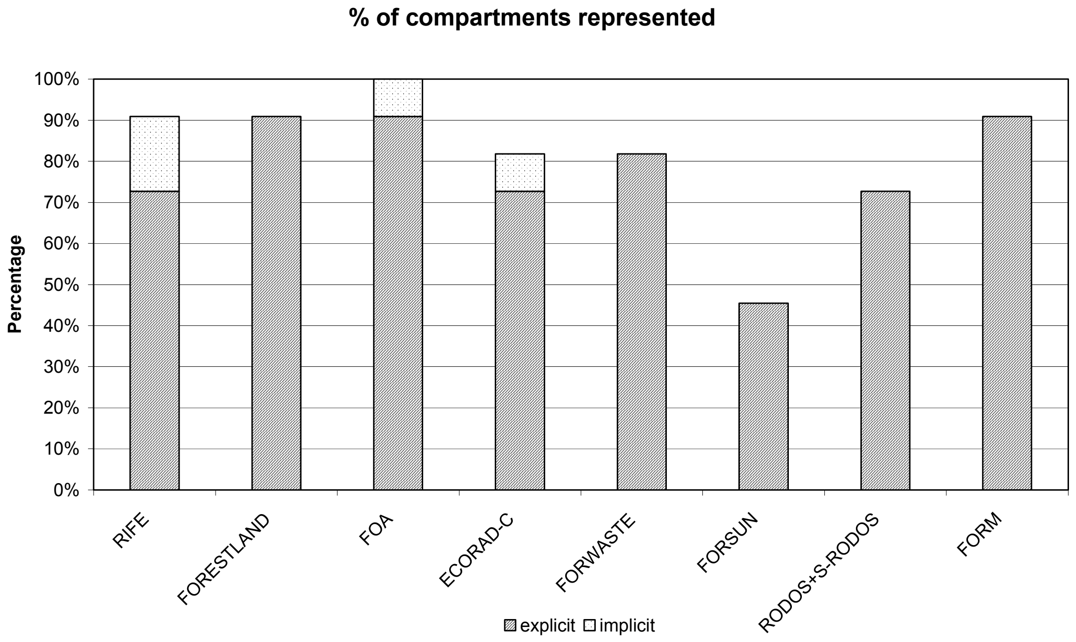


FIG. 3. Explicit/implicit model representation of forest ecosystem compartments listed in the Forest Matrix (Figure 2).

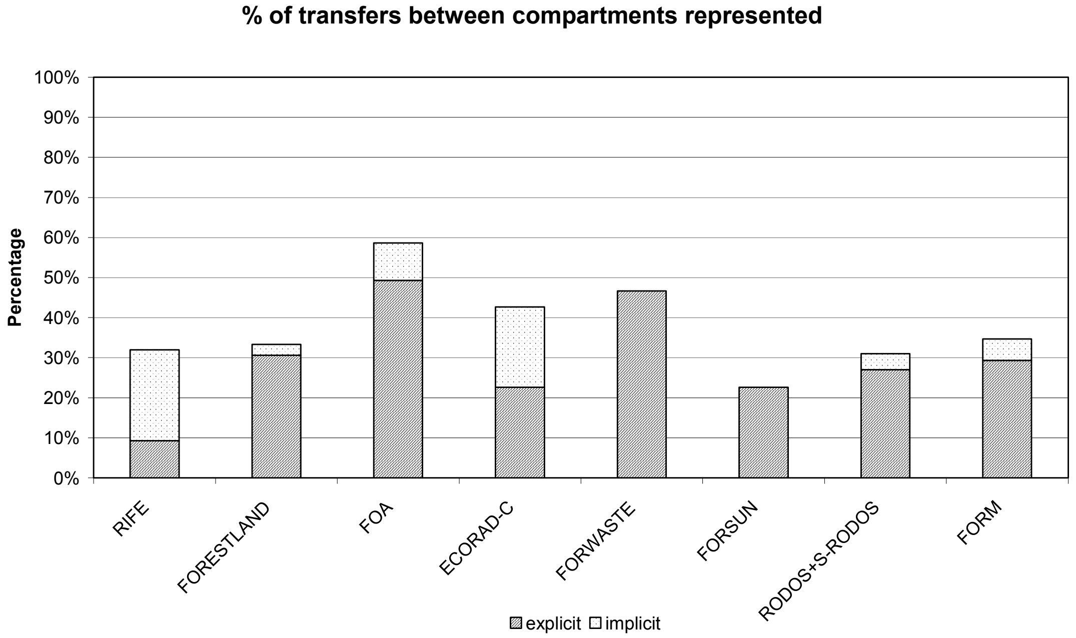


FIG. 4. Explicit/implicit model representation of transfers between forest ecosystem compartments listed in the Forest Matrix (Figure 2)[2].

---

[2] Models such as FORSUN and RODOS were still under development during this study.

3. DEFINITIONS OF TRANSFER PARAMETERS FOR UNDERSTOREY VEGETATION AND FUNGAL FRUIT BODIES

3.1. INTRODUCTION

The quantitative evaluation of radioactivity levels in fungal fruit bodies and understorey vegetation is a challenging task. The concept of transfer factors and concentration ratios, commonly used in agricultural radioecology, has also been applied extensively to quantify the transfer of radionuclides from soil to fungi and understorey vegetation in non-agricultural situations. This concept has been recommended by several agencies: IAEA [1994], IUR [1992] and ICRP [1979]. In general, transfer factors are expressed as parameters representing the contamination (activity concentration) of green plants or fungal fruit bodies divided by parameters representing the contamination (activity concentration or inventory) of soil. It is implicitly assumed that the radionuclide activity concentration in plants or fungal fruit bodies can be described by a linear function of soil contamination passing through the origin. The contamination of plants and fruit bodies is usually expressed as the amount of radioactivity per unit weight, either on a dry weight (Bq kg^{-1} dry weight) or a fresh weight basis (Bq kg^{-1} fresh weight). The contamination of soil is usually expressed as the amount of radioactivity per unit surface area (i.e. an inventory, Bq m^{-2}) or per unit dry weight (Bq kg^{-1} dry weight), where the latter definition commonly refers to standardised soil depths.

Until now, there has been no concrete evidence as to whether transfer parameters should refer to the total radionuclide inventory in soil, or the bioavailable part of the activity, or the activity concentration within a certain soil depth or, specifically, the activity concentration of the soil layers exploited by fungal mycelia or fine roots. In general, there is no ‘best definition’ for transfer parameters, since these empirical parameters are often intended to be used in different ways in particular radioecological models. Consequently, the suitability of a certain definition of transfer parameters depends on the purpose of the radioecological model and the knowledge about the ecosystem to be modelled. If, for example, activity levels have to be estimated very quickly after an accidental release of radionuclides, aggregated transfer factors, which refer to the total deposition on the ground surface, might be a good choice to assess the order of magnitude of contamination levels to be expected in fungal fruit bodies and green plants. For longer-term predictions of activity levels, however, more sophisticated definitions of transfer parameters might be more suitable. It is commonly found that concentration ratios explicitly referring to the soil horizons within which fungal mycelium or fine roots are located, work well, notably for sampling sites with a significant vertical migration of radionuclides in the soil.

Despite the conceptual simplicity of transfer parameters, their inherent limitations should always be acknowledged and taken into account to avoid erroneous applications of these parameters and wrong conclusions concerning the availability of radionuclides for uptake by green plants and fungi. It is generally accepted that the large observed variations of radionuclide levels in fungi and green plants cannot be entirely explained by varying concentrations in the soil alone. The radionuclide activity concentrations in fungi and green plants depend also on other ecological parameters, such as the type of soil, its content and type of clay minerals, etc. In addition it is well known that green plants have the ability to control the uptake of ions, thus regulating their content of essential elements. Wirth et al. [1985] argue that there are only slight variations in the concentrations of essential elements in green plants and, therefore, the variances of the corresponding transfer factors mainly represent the concentration variance of the essential elements in soil. In this case, the naive application of transfer factors or concentration ratios would lead to values that decrease with increasing soil content. McGee et al. (1996) conclude that such a decreasing concentration ratio value is a mathematical artefact that arises as a result of dividing a denominator (soil concentration) which shows considerable variation, into a numerator (plant concentration) which is relatively constant. A critical review of the concept of ratios in the field of radioecology was published by McGee et al. [1996].

A considerable lack of knowledge exists concerning the mechanisms and processes involved in radionuclide uptake and retention by fungi and green plants which, in forest soils, often live in close symbiotic relationships. It is therefore not surprising that transfer factors and concentration ratios are a popular empirical parameter to describe quantitatively the uptake of radionuclides from soil to fungi and green plants, notably in forest ecosystems.

3.2. DEFINITIONS OF TRANSFER FACTORS

This section presents common definitions of transfer factors for soil–green plant transfer and soil–fungi transfer of radionuclides. The specific advantages and limitations of different definitions are briefly discussed.

3.2.1. Aggregated transfer factors

Aggregated transfer factors (often referred to as T_{agg} values) are defined as the ratio of the activity in plant (Bq kg^{-1} fresh weight or Bq kg^{-1} dry weight) divided by the total deposition on the soil (Bq m^{-2}). Sometimes, other area-related definitions of transfer factors, e.g. area-related transfer factors referring to a certain soil depth instead of the total deposition, are also called aggregated transfer factors, a fact that might lead to confusion.

The concept of aggregated transfer factors was developed, *inter alia*, to avoid difficulties in defining a suitable average radionuclide activity concentration in soils with a multi-layered structure, which are common in many natural and semi-natural environments. Aggregated transfer factors are a useful tool to estimate quickly, but only approximately, the uptake of radionuclides by vegetation, often during the short-term following accidental release of radionuclides but also over the long term (i.e. decades). However, aggregated transfer factors suffer from some disadvantages in connection with radioecological models. Since they refer to the total deposition on soil, they usually exhibit a time-dependence: this effect is particularly pronounced for plants and fungal species which exploit distinct soil horizons, especially when significant vertical migration of radionuclides occurs within the soil. A wide range of aggregated transfer factors (more than four orders of magnitude) has been reported for fungi. As will be discussed later, this large variation is, at least partially, attributable to non-uniform vertical distributions of radionuclides in the soil and the fact that fungal mycelia often colonise distinct soil horizons.

3.2.2. Transfer factors (concentration ratios) referring to standardized soil depths

Transfer factors referring to standardised soil depths are defined as the ratio of the activity concentration in plant (Bq kg^{-1} fresh weight or Bq kg^{-1} dry weight) divided by the activity concentration in soil (Bq kg^{-1} dry weight) within the uppermost layer of a standardised soil thickness. This definition was designed especially for agricultural ecosystems, where radionuclides are often distributed homogeneously within the rooting depth of agricultural plants due to ploughing.

As in the case of aggregated transfer factors, transfer factors defined on the basis of standardised soil depths are of limited usefulness in the case of soils with a multi-layered structure and a pronounced vertical profile of activity concentration. Averaging the radionuclide concentration over a standardised soil depth, irrespective of the location of the mycelium or the fine roots, might lead to a large variation and a time dependence of transfer factors. Details will be discussed in Section 3.2.4.

3.2.3. Transfer factors soil solution – plant

Several studies [Horrill et al. 1990; Desmet et al. 1991; Schell et al. 1996, Mytennaere et al. 1993 and Tikhomirov et al. 1993) suggested that the bioavailability of a given radionuclide in soil, not its bulk concentration, is important for uptake by fungi and green plants. Expressing transfer parameters on a dry soil basis has been criticised as being inadequate. Desmet et al. [1991] argue that the metal ion concentration and its chemical form in soil solution are the most important factors in determining the root uptake and transfer to plant tissues. Consequently, the possibility to calculate transfer factors as the ratio of the activity concentration in plant (Bq kg^{-1} fresh weight or Bq kg^{-1} dry weight) divided by the activity concentration in soil solution (Bq l^{-1}) has been discussed. Nevertheless, Desmet et al. [1991] state that long-term bioavailability relates to the more strongly binding organic complexes and minerals and to the solid phase of the soil. In this context, the available fraction has been defined qualitatively as the fraction of an element that has the potential to move into the soil solution.

Experimental results for forest soils have revealed that the fraction of easily exchangeable radiocaesium in organic horizons is low but, nevertheless, radiocaesium is highly available for uptake by fungi. This effect can be very pronounced for organic horizons of forest soils. At present, there is no experimental method available to quantify the bioavailable fraction of radionuclides in soil under semi-natural conditions. The results of field studies indicate that most radiocaesium in the organic horizons may be available for uptake by fungi. Rühm et al. [1999] reported that the bioavailabilities of stable ^{133}Cs and radioactive ^{137}Cs and ^{134}Cs are approximately the same in the organic horizons of a German forest site.

3.2.4. Transfer factors relating to specific soil horizons

Transfer factors (concentration ratios), defined as the ratio of the activity concentration in fungal fruit bodies or green plants (Bq kg^{-1} fresh weight or Bq kg^{-1} dry weight) divided by the activity concentration of the specific soil layer exploited by the mycelium or the root system (Bq kg^{-1} dry weight) have proved to be useful, especially for application in dynamic radioecological models. This definition of the transfer factor was proposed in the late 1980s.

It is very difficult to determine the vertical location of fungal mycelium within soils on a species-by-species basis. In the case of radiocaesium, Byrne [1988] and Guillitte et al. [1990] suggested that the isotopic ratio of ^{134}Cs/^{137}Cs could be used for this purpose. The approach is based on the idea that the isotopic ratio in fungal fruit bodies should reflect the isotopic ratio of the soil horizon from which radiocaesium is predominantly taken up. At several sampling sites the time-dependent isotopic ratio ^{137}Cs/^{134}Cs has turned out to be a 'fingerprint' of the different layers of forest soil, a consequence of the mixing of the residual ^{137}Cs from global fallout from atmospheric nuclear tests with ^{134}Cs and ^{137}Cs from the Chernobyl fallout. Hence, the location of fungal mycelia in forest soils can be determined by comparing the isotopic ratios of ^{137}Cs/^{134}Cs in fruit bodies with the corresponding values of different soil horizons. This basic idea was developed in an operational tool by Rühm et al. [1997] who determined the vertical location of the mycelia of 14 fungal species in German forests.

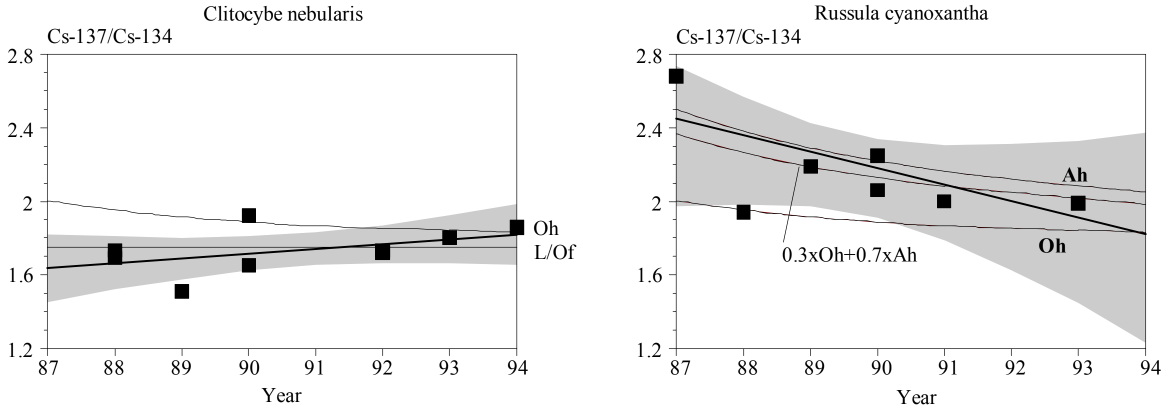


FIG. 5. ^{137}Cs/^{134}Cs ratios as functions of time in two mushroom species at a German sampling site, decay-corrected to May 1, 1986. The black squares denote measurements. The thick full line represents linear regression curves and the shaded areas are the corresponding 95 percent confidence bands. The predictions of a compartment model for the isotopic ratios of different horizons are shown for comparison.

As an illustration, Figure 5 presents the isotopic ratios of ^{137}Cs/^{134}Cs as functions of time measured in samples of *Clitocybe nebularis* and *Russula cyanoxantha,* together with 95 percent confidence bands. The ^{137}Cs/^{134}Cs ratios for different soil horizons are also shown for comparison. Obviously, *Clitocybe nebularis* has a superficial mycelium located in the L and/or Of horizon. The ^{137}Cs/^{134}Cs ratios in samples of *Russula cyanoxantha* are significantly higher and indicate that this symbiotic species obtains radiocaesium from both Oh and Ah horizons.

The advantage of transfer factors referring to specific soil horizons, from which nutrients and radionuclides are mainly taken up, is three-fold:

(1) The variation of transfer factors for specific horizons is significantly lower compared with other definitions, notably in the case of a non-uniform vertical distribution of radionuclides in soil. Guillitte et al. [1990] even conclude that, in the case of fungi, “soil sampling at a constant depth has no practical interest when the element is unequally distributed across the soil profile”.

(2) Transfer factors for specific horizons offer a conceptual advantage for predictive modelling. Transfer factors defined in this way have not changed significantly for the period starting several years after the Chernobyl accident [Rühm et al. 1998] and there is evidence that they will stay fairly constant in future [Rühm et al. 1999].

(3) Transfer factors for specific horizons are a direct measure of the availability of radionuclides for uptake by fungi or green plants.

The methodology described above is very difficult to apply now, since ^{134}Cs is a short-lived radionuclide. Currently, research projects are dealing with the question of whether the ratio ^{137}Cs/stable caesium can alternatively be used to localise fungal mycelia or fine roots of green plants *in situ*, an approach which is expected to work well for organic horizons of forest soils. In organic soil layers, where the fraction of caesium fixed within mineral particles is likely to be very small, the ratios of radiocaesium/stable caesium in fruit bodies were close to those of the soil layers, from which certain species of fungi take up radiocaesium [Rühm et al. 1999]. If mycelia colonise deeper soil horizons, where the concentration of mineral particles is usually increased, the ratios of radiocaesium/stable caesium in fruit bodies can be higher compared with the ratio in the corresponding soil layer [Tsukada et al. 1998; Yoshida et al. 1998]. This trend is to be expected if a significant fraction of stable caesium is enclosed in mineral particles and thus not available for uptake by fungi.

3.2.5. Rhizosphere–plant transfer factors

Based on the results of a recent study, Delvaux et al. [2000] proposed transfer factors that refer to the rhizosphere, i.e. the volume of soil influenced by plant root activity. Plant roots are "dynamic weathering agents" which strongly modify their soil environment while taking up their nutrients. The uptake of potassium, for example, induces potassium depletion around plant roots and hence a rapid weathering of mica through the release of interlayer potassium. Delvaux et al. [2000] argue that the mobilisation of otherwise unavailable radiocaesium might be directly linked to this process.

The concept of rhizosphere – plant transfer factors under extreme potassium deficiency has been tested with pot experiments, which were specifically designed to investigate the rhizospheric effects of radiocaesium uptake [Delvaux et al. 2000]. A root mat of ryegrass was brought into close contact with a soil–agar mixture spiked with carrier-free ^{137}Cs. The plants were supplied with a nutrient solution free of potassium to simulate the potassium depletion in the rhizosphere of unfertilised soil. The rhizosphere – plant transfer factor could not be related to any common physical or chemical soil property but was significantly correlated with the radiocaesium interception potential (RIP). Thus, the RIP quantitatively relates the rhizospheric mobilisation of radiocaesium with an intrinsic caesium binding property of soils.

Up until now, the concept of rhizospheric transfer factors has not been applied to field investigations. Moreover, biological interactions, such as the effects of mycorrhizal fungi, which occur particularly in the upper horizons of forest soil, have not yet been considered.

3.2.6. Time-dependence of transfer parameters

The derivation and application of transfer factors rely on the assumption that radionuclide distribution in a plant, an animal or a fungal fruiting body has achieved equilibrium. However, radionuclide redistribution in soil after acute deposition, notably vertical migration in the soil, generally leads to the problem that transfer factors are time-dependent if definitions referring to the mean activity concentration within a standardised soil depth or the total inventory are used. Transfer factors defined in this way are expected to peak when the maximum of the vertical radionuclide distribution reaches the rooting zone or the soil layer with maximal mycelium density. Many experiments have been designed to study these temporal changes, to tabulate time-dependent transfer factors and to use them for modelling. Recently, Gillett and Crout (2000) demonstrated statistical evidence for the time dependency of aggregated transfer factors, based on an extensive literature review and experimental data sets, although individual studies often report different time series for different species.

For predictive modelling it is highly desirable to use a radioecological model with the dynamics conceptually included in the model itself and model parameters constant over time. This statement is also true for transfer parameters. Those definitions of transfer factors which are not constant, but depend on time in some unpredictable way, should be applied very carefully if the long-term contamination of fungal fruit bodies and green plants is to be modelled successfully. Thus, transfer factors such as T_{agg} might not be appropriate for long-term predictions since their detailed change over time is generally unknown.

For long-term predictions of contamination levels in fungal fruit bodies and green plants, transfer parameters which explicitly refer to the soil layers from which nutrients and radionuclides are taken up may offer a conceptual advantage. A time series of field measurements at a German forest site revealed that transfer factors defined in this way did not change significantly over a period of several years following the Chernobyl accident, though a possible time dependence could not entirely be ruled out (Rühm et al., 1998). This study found that changes in activity levels in fungal fruit bodies and understorey vegetation were due to physical decay or migration of radiocaesium to deeper soil layers.

3.3. CONCLUSIONS

The concepts of transfer factors and concentration ratios provide a popular approach to quantify the transfer of radionuclides from soils to plants and fungal fruit bodies. This chapter has reviewed common definitions of transfer factors and highlighted their advantages and limitations. It has been shown that there is no 'best choice' of any particular type of transfer parameters since, in most cases, these empirical parameters do not stand alone but belong to a specific radioecological model. Consequently, the suitability of a particular definition of a transfer parameter depends on the purpose of the radioecological model, the manner in which the data to calculate the transfer parameter have been obtained and the level of understanding of the ecosystem to be modelled. The latter is often limited by the availability of field data, either obtained directly by measurement or from the literature.

In the past, numerous values for transfer factors have been published in the literature. In many cases, however, the authors did not specify the precise definition of the transfer factors they used. This might lead to confusion and erroneous application of such transfer factors. Area-related transfer factors, for example, are often called 'aggregated transfer factors', even if they do not refer to the total inventory of radionuclides within the soil. It is strongly recommended that any published transfer factor values are accompanied by a detailed description of the way in which the values were obtained and how they should be used to calculate soil-to-plant or soil-to-fruit body transfer.

This section has been exclusively concerned with the definition and application of transfer factors for herbaceous vegetation and fungal fruiting bodies. Quantification of the uptake of radiocaesium by trees for the purposes of predictive modelling has been given very little consideration in comparison with these vegetation types. This problem is addressed in Section 4.

4. THE ADEQUACY OF TRANSFER FACTORS TO ESTIMATE LONG TERM ACCUMULATION OF RADIOCAESIUM IN WOOD

4.1. INTRODUCTION

In forests affected by Chernobyl fallout, long-term management of ^{137}Cs-contaminated wood stocks requires a reliable tool to estimate the evolution of the ^{137}Cs content in timber wood in the future [Ipatyev et al. 1999]. In the long term, root uptake is the dominant source of ^{137}Cs contamination in growing wood. As for other radionuclides, ^{137}Cs contamination in forest vegetation is generally assessed with transfer factor (TF) or aggregated transfer factor (T_{agg}) coefficients that express the ratio of the average radionuclide concentration in a plant compartment (Bq kg^{-1}) to that in soil (Bq kg^{-1} for TF or Bq m^{-2} for T_{agg}). Depending on the time elapsed after an atmospheric pulse, it is not possible to distinguish whether the measured radiocaesium contained in the wood compartment originated from initial atmospheric deposits or from root uptake [Ertel and Ziegler 1991; Bonnett and Anderson 1993; Fawaris and Johanson, 1994; Barci-Funel et al. 1995; Haas et al. 1995]. TFs refer to the total content of ^{137}Cs in wood and can therefore not be used to distinguish between the respective contributions of each process. The adequacy of TFs to predict future ^{137}Cs accumulation in wood due to root uptake is therefore questionable, as clearly illustrated by comparing transfer factor values with the real ^{137}Cs stock in wood in two neighbouring contaminated pine (*Pinus sylvestris L.*) stands (17 and 58 years old, respectively) located near Gomel in Belarus and contaminated with similar ^{137}Cs deposits (Table 2).

TABLE 2. COMPARISON OF TF WITH ^{137}Cs CONTENT IN WOOD IN PINE PLANTATIONS (VETKA, BELARUS)

	17 years	58 years
Trunk biomass (tonne ha^{-1})	37.03	152.27
Transfer factor (Tagg)(m^2 kg^{-1}) – measured	0.0047 (0.0013)	0.0035 (0.0008)
^{137}Cs content in soil (kBq m^{-2})	1462.1 (116.7)	1103.9 (228.3)
^{137}Cs content in trunk wood		
calculated (% of soil activity)	1.74 (0.14)	5.33 (1.10)
measured (% of soil activity)	1.52 (0.35)	8.58 (0.31)

With increasing age, measured TF values decrease while ^{137}Cs total content in trunk wood is greatly enhanced. Moreover, the increase in standing wood biomass can only partly explain the discrepancy in ^{137}Cs amounts accumulated in wood between young and old stands. This indicates that historical accumulation of ^{137}Cs in wood does not simply relate to root uptake as the main vector of wood contamination. Initial interception and subsequent incorporation of ^{137}Cs were likely very important in older trees.

Radiocaesium is indeed highly mobile and rapidly recycled in trees. Stemwood acts as an important reservoir of ^{137}Cs but its radial distribution between rings is not conservative. However, the observed radial gradient of ^{137}Cs reflects a particular diffusion process in trunk biomass. The radial pattern of ^{137}Cs thus poses a problem when using the average level of ^{137}Cs in wood and derived TF coefficients for a prospective estimation of further ^{137}Cs accumulation in wood. A new estimate of the radiocaesium immobilization, i.e. the wood immobilization potential (WIP), was therefore introduced and is described below.

4.2. THE WOOD IMMOBILIZATION POTENTIAL

The definition of WIP is based on the shape of the accumulated curve shown schematically in Figure 6 for radiocaesium (plain curve). This curve was plotted by adding, for each annual wood increment from the pith to the last formed ring, respectively, the volume of trunk wood (cm^3) and its radiocaesium content (Bq) as x- and y-axes (the point 0 corresponds to the present situation). The shape of this curve, based on ^{137}Cs levels measured more than ten years after the initial contamination pulse, will probably not evolve drastically in the near future, and the newly incorporated ^{137}Cs (which only originates from root absorption at this time) will be redistributed accordingly. The WIP, defined as the limit of the mathematical slope of the curve in point 0 (curve 3), can consequently be considered as a suitable estimate of the future net ^{137}Cs accumulation in wood due to root uptake. The WIP is therefore a compromise between two extreme approaches: a conservative approach (each ring preserves the memory of the annual uptake episode, i.e. the ^{137}Cs is not redistributed in the trunk – curve 1) and the TF approach (there is no radial gradient of ^{137}Cs assumed to be homogeneously redistributed in the trunk – curve 2).

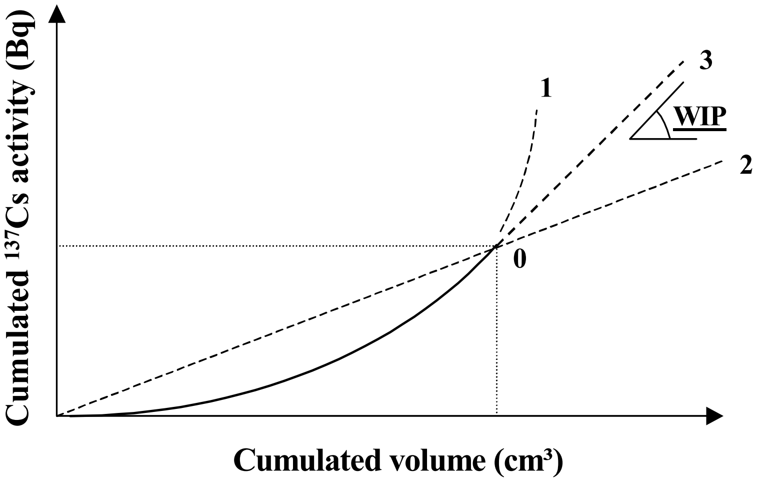


FIG. 6. Conceptual definition of the wood immobilization potential (WIP).

The comparison of WIP with transfer factors (Table 3) shows that WIP values increase as the tree ages, contrary to what is suggested by transfer factors.

TABLE 3. COMPARISON OF WIP WITH OTHER WAYS TO CALCULATE ^{137}Cs ACCUMULATION IN WOOD

Immobilization	17 years	58 years
Measured		
Transfer factor (m^2/kg^{-1})	0.0047 ± 0.0013	0.0035 ± 0.0008
Total Cs content (wood) (% soil activity)	1.52 ± 0.35	8.58 ± 0.31
Calculated		
Cs WIP (Bq cm^{-3}y^{-1})	1.99 ± 0.30	3.12 ± 0.23
Annual Cs flux to wood (% of soil activity)	0.15 ± 0.04	0.34 ± 0.07
Cs content in wood due to root uptake (% of soil activity)	1.80 ± 0.48	4.08 ± 0.84

Contrary to what is shown for TF values, the WIP for ^{137}Cs increases as the tree ages. This means that, at present, old trees immobilise about 1.5-fold more ^{137}Cs per unit volume of newly formed wood than the younger tree via root uptake. The WIP approach allows us to estimate the current annual ^{137}Cs flux to wood by multiplying WIP values with the current annual wood increment of the stand, reported to soil activity. The annual ^{137}Cs flux to wood, multiplied by the time elapsed since the Chernobyl accident (12 years), gives an idea of the current average ^{137}Cs content due to root uptake. This calculated value is in good agreement with the measured total ^{137}Cs content in trunk wood of the young stand that was, in fact, only slightly affected by initial interception. For the old stand, the same calculation shows that soil-to-wood transfer explains only 48% of the measured total ^{137}Cs content in wood. This is consistent with the previous assumption. Finally, the root uptake of ^{137}Cs in a forest soil depends on the contamination level, the ^{137}Cs bio-availability and the distribution of roots in the different layers. In the older stand (58 years), the higher WIP was associated with a higher accessibility of the ^{137}Cs located in the organic layers (data not shown).

5. MODEL–MODEL INTER-COMPARISON STUDY – ‘SCENARIO 1’

5.1. INTRODUCTION

This section describes results from the first model inter-comparison study undertaken by the BIOMASS Forest Working Group. The study involved comparison of outputs from 10 different models (see Section 5.3). The contamination scenario (‘Scenario 1’) that was posed for this exercise was hypothetical, but based on data sets from a real forest. It was deliberately made simple in order to allow all potential contributors of model predictions to provide results with the minimum of difficulty, thus ensuring a high rate of return of results from modellers. The scenario, summarised in Section 5.2, was based on a Chernobyl-type pulse input to a coniferous forest ecosystem of which all modellers were likely to have experience and appropriate model calibrations. Since Scenario 1 was hypothetical, the aim of the inter-comparison was not to validate model predictions against actual data but to compare the results produced by each modeller and his/her respective model. This model–model comparison was intended to serve as a baseline against which future scenarios and model–data validations could be gauged. Thus, one of the primary aims of the exercise was to allow modellers to assess the need to modify their model approach before undertaking further exercises within the Forest WG.

5.2. SUMMARY OF SCENARIO 1

The complete scenario is listed in Annex II. The scenario was based on a hypothetical Chernobyl-type pulse input to a coniferous forest ecosystem although the description of the forest ecosystem was based on real data from a coniferous forest typical of or south-western regions of Russia. The main soil type is a soddy-podzolic loamy sand. The dominant tree species is Scots pine (*Pinus sylvestris*), with some birch (*Betula pendula*). The understorey includes red raspberry (*Rubus idaeus*) and blackberry (*Rubus trivialis*), as well as mushroom species such as *Boletus edulis*, *Leccinum scabrum*, *Cantharellus cibarius* and *Russula* species. Grasses are rather sparse, but mosses cover 90% of the ground area. The main game species are moose and roe deer.

Participants were requested to predict activity concentrations of ^{137}Cs on a fresh weight basis. Preferred endpoints could be chosen from the following list:

- total tree;
- total wood (i.e. trunk plus branches);
- needles (annual average);
- other parts of tree, especially bark;
- soil profile, including litter;
- animals (annual average for moose and roe deer); and
- vegetation (‘mushrooms’, berries, shrubs and grass).

Scots pine (*Pinus sylvestris*) was the main tree of interest, but participants could report on both pine and birch (*Betula pendula*) if they wished. Similarly, participants could report generically on mushrooms, berries, shrubs and grasses, or on particular species of these.

Each chosen endpoint was to be considered as a function of time and results were requested at 1 year intervals from 1 to 20 years after the date of initial deposition. It was requested that calculations be based on best estimates of parameter input values and that results be reported as both ‘best estimates’ and/or 95% confidence intervals, if possible.

5.3. PARTICIPANTS AND MODELS

Individual modellers and their respective models are listed in Table 1. Brief descriptions of each of these models are provided in Annex I.

5.4. RESULTS

The results are presented graphically as 20 year time courses following initial deposition. As requested in the scenario sent out to respondents, the time resolution of these time courses is 1 year.

All the modellers reported results as ‘best estimates’. The results obtained using the FORESTPATH model were also reported as 95% confidence intervals. To simplify the graphs, normally only the median value calculated with the FORESTPATH model is plotted. The 95% confidence intervals are only plotted when it differs significantly from the median value.

5.4.1. Trees and associated components

Seven respondents provided predictions for either total tree activity concentrations and/or activity concentrations of specific components of trees. Figures 7–10 show plotted results for the specific tree-related endpoints requested, namely total tree (all tissues averaged), wood (trunk plus branches), needles and bark, respectively.

In the case of total tree, bark and needle predictions the relative uniformity of predicted time courses after approximately 5 years was striking. The FOA model consistently predicted the highest activity concentration within the total tree (Figure 7), a result which is accounted for by the relatively high activity concentration in wood predicted by this model (Figure 8). The range of predicted activity concentrations in total tree tissues was greatest at 0 years and 20 years (i.e. at the very beginning and end of the simulation period) although this maximum range was just greater than one order of magnitude. A similar maximum range of values for activity concentrations in wood was observed. Five modellers returned predictions for wood and within this group of predictions two distinct types of predicted time course emerged. In the case of FOA, RIFE and FORESTLIFE the predicted time courses were characterised by an initial increase in wood activity concentration to a maximum (at times ranging from approximately 4 to 8 years following deposition) after which the activity concentration declined. Somewhat different time courses were predicted by FORESTLAND and LOGNAT, which showed a general increase in wood activity concentration over the 20-year simulation period.

As with predictions for wood, the predicted time courses of activity concentrations in bark could be separated into two sub-groups. Of the four models that provided bark predictions the FORESTLAND and ECORAD models predicted approximately exponentially declining time courses while RIFE and FORESTLIFE predicted rather lower initial bark activity concentrations and rather higher activity concentrations in bark after approximately nine years (Figure 10). This is probably indicative of different conceptual approaches in each of these two pairs of models. For instance, in FORESTLAND bark is defined as the outer surface of the tree, whereas the other models do consider the internal bark to a certain extent.

The maximum range of results for bark was large with a range of approximately three orders of magnitude being spanned by predictions at 20 years – this was largely accounted for by the exponentially decaying time course predicted by FORESTLAND.

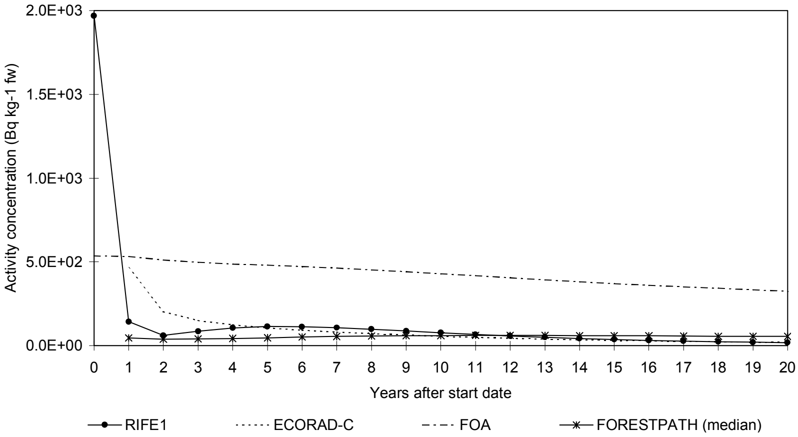


FIG. 7. ^{137}Cs activity concentration in total tree (Bq kg^{-1} fresh weight).

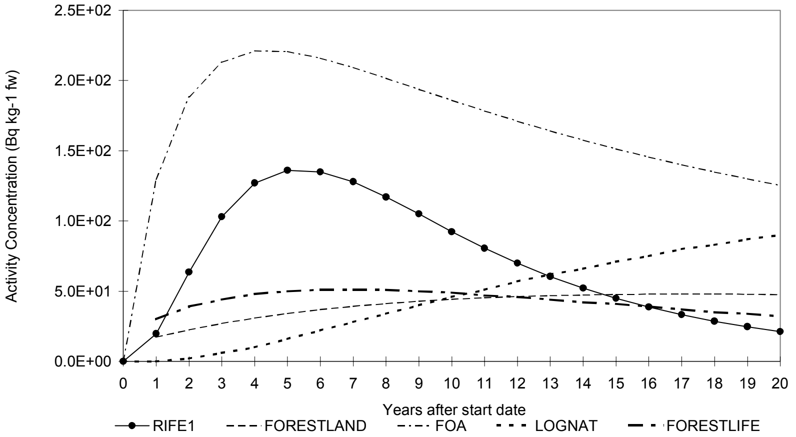


FIG. 8. ^{137}Cs activity concentration in pine wood (trunk+ branches) (Bq kg^{-1} fresh weight).

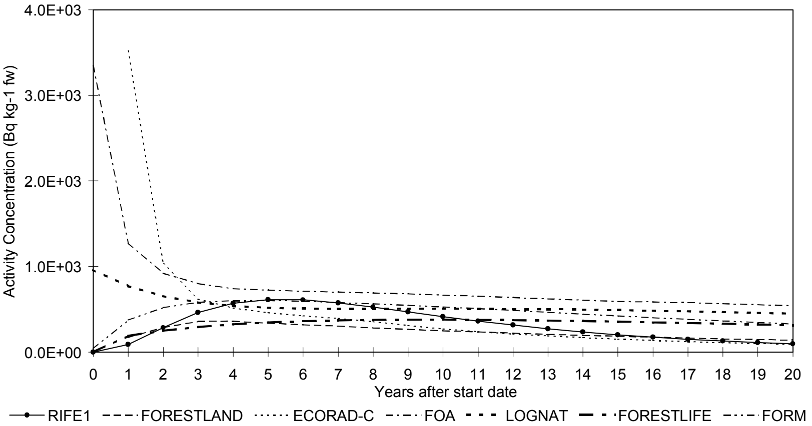


FIG. 9. ^{137}Cs activity concentration in pine needles (Bq kg^{-1} fresh weight).

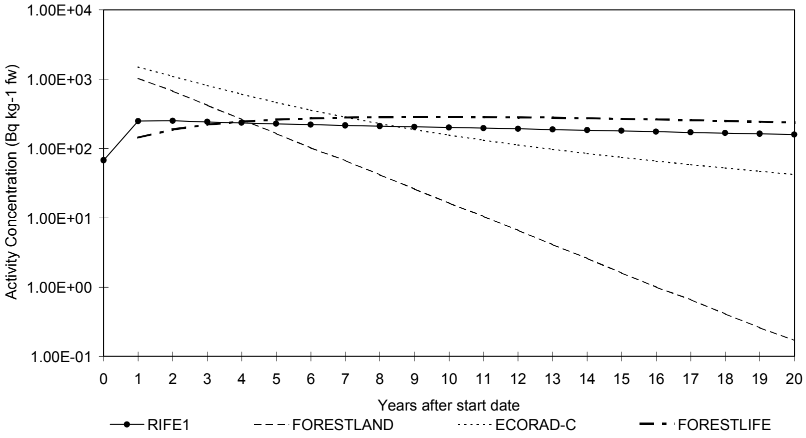


FIG. 10. ^{137}Cs activity concentration in pine bark (Bq kg^{-1} fresh weight).

Predicted time courses of activity concentrations in needles (Figure 9) show a remarkable degree of consistency after three years with a maximum range of about one order of magnitude at 20 years. In the initial three years of the predictions, however, both ECORAD and FOA predicted time courses of needle activity concentrations rapidly declining from initially high values. LOGNAT predicted a similar though less pronounced time course. FORESTLAND starts simulating the activity in needles one year after deposition only, as it is assumed that most of the intercepted activity is released during the first year (this assumption is made for long-term calculations only). The other models (RIFE and FORESTLIFE) predict initially zero needle activity concentrations followed by a steady increase over a period of three to ten years. The discrepancy between the models points unequivocally to a difference in interpretation of the desired endpoint by the modellers. In the case of ECORAD, FOA and LOGNAT it is clear that initial external contamination of needles, due to direct deposition from the atmosphere, is taken into account whereas in RIFE, and FORESTLIFE only contamination via root uptake and subsequent internal translocation is considered.

5.4.2. Soils

Soils are subdivided into three major layers for the purposes of the analysis of results from the model–model inter-comparison, namely litter (Figure 11), organic soil (Figure 12) and mineral soil (Figure 13). Interpretation of the actual soil horizons which constitute each of these broad layers can vary between modellers but, broadly, the litter layer is taken to be the AoL horizon, the organic layer is taken to be the AoF plus AoH horizons and the mineral soil is taken to be anything below the organic horizons (on the basis of this definition the mineral soil may, in practice, include hemi-organic horizons marking the transition between organic and mineral layers).

As for the prediction of contamination of trees, seven modellers returned predictions of contamination time courses for soils. There was generally good quantitative consistency between predicted time courses for all three major soil layers with maximum variation between predictions never exceeding slightly more than one order of magnitude. Qualitatively, too, the predicted time courses were very consistent with each other, especially in the case of the mineral soil predictions (Figure 13). This probably represents a high degree of consistency in the way in which soil migration is represented conceptually within each model. In most of the models a classical compartmental leaching approach is taken in which downwards migration of radiocaesium occurs from one discrete compartment to the next. The FORESTLIFE model represents this process by means of a physical advection/diffusion approach, yet the predictions of this model are very similar to those of the other models. The FOA model distinguishes between two soil compartments “available soil” and “unavailable soil”, which are not related to position in any soil layers. However, ”available soil” is expected to correspond mainly with the organic layers below the AoL horizon, and “unavailable soil” to the mineral soil, allowing for exchange of caesium between the “unavailable” condition to the “available”.

The particularly high degree of consistency between predictions of contamination of soil layers doubtless reflects the strong effort which has been put into understanding soil migration, not only in forests but particularly in agricultural ecosystems.

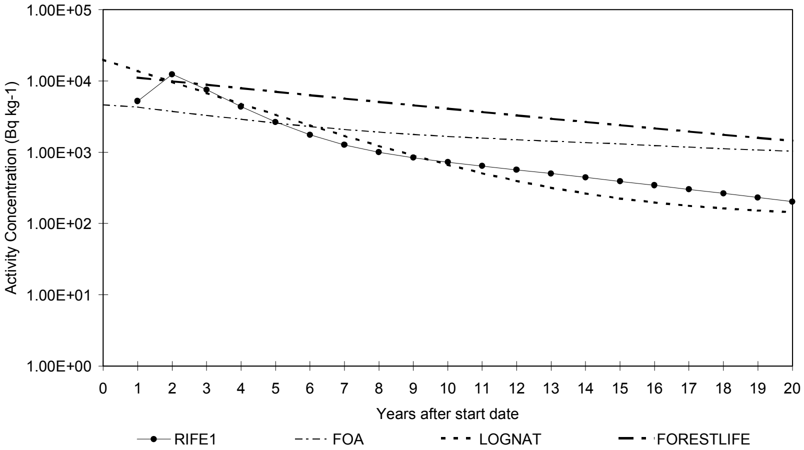


FIG. 11. ^{137}Cs activity concentration in soil litter layer (Bq kg^{-1}).

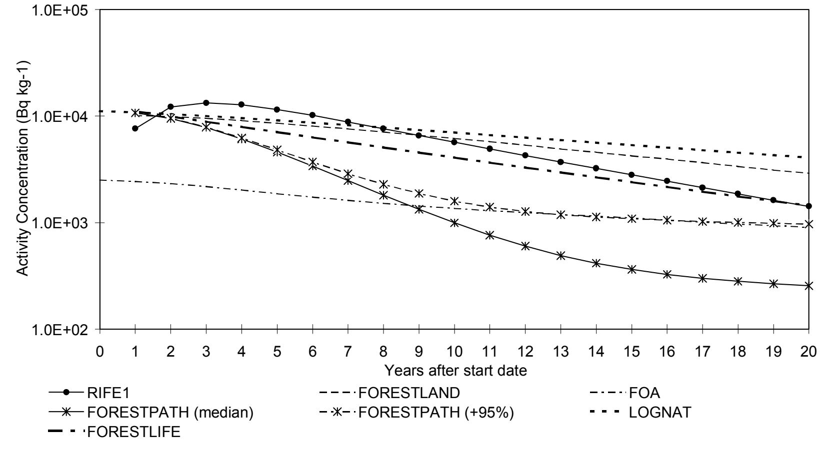


FIG. 12. ^{137}Cs activity concentration in organic soil layer (Bq kg^{-1}).

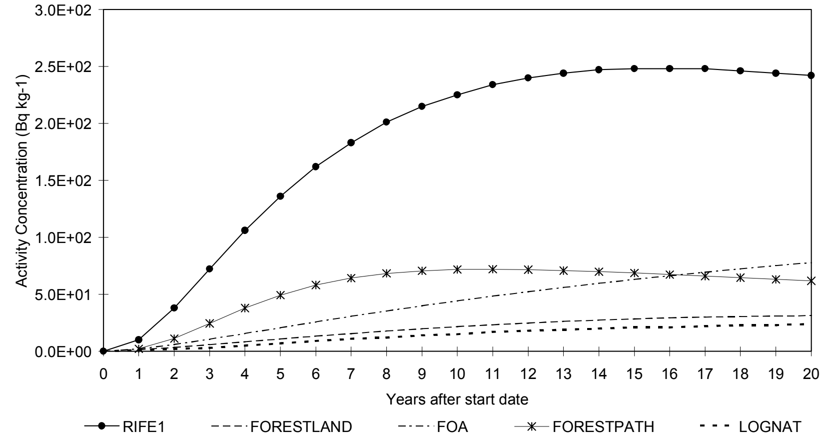


FIG. 13. ^{137}*Cs activity concentration in mineral soil layer (Bq kg*$^{-1}$*).*

5.4.3. Other biological endpoints

The biological endpoints (other than trees) specified in Scenario 1 are of concern because their consumption by man represents a radiological exposure route via which internal doses may be incurred. Thus, wild animals, understorey (berries) and mushrooms are the main biological endpoints of interest in this study – model predictions of contamination time courses for each of these components of the forest ecosystem are shown in Figures 14–19, respectively.

5.4.3.1. Wild animals

Five modellers provided predictions for time courses of contamination in wild animals (game). Of the five models, one (RIFE) interprets game in a generic sense while the remaining four specify the species of interest, namely deer and/or moose. The spread of predictions produced by these models appears large in Figure 14 (moose) and Figure 15 (deer) but is generally within one order of magnitude. The time trends predicted for moose, deer and generic 'game' are all qualitatively similar, with similar predictions of gently declining activity concentration over the 20 year simulation period. There are striking quantitative differences between predictions for moose and deer contamination, with the greatest variability being evident in predicted moose time courses. The differences between the uppermost and lowermost moose contamination predictions (FORESTLAND and FINNFOOD, respectively) was approximately a factor of three, although there was very good agreement between these two models for predictions of deer contamination. Finally, the agreement between the RIFE model (using a generic, IAEA recommended, T_{agg} value) and the FOA model (which provided a prediction for moose only) was striking, particularly so since the FOA moose model is based on seasonally dependent intake rates from vegetation compartments, and does not use the T_{agg} principle.

5.4.3.2. Understorey

Predictions of time courses of contamination of the understorey focussed on berries (Figure 16). Results were provided by four of the modellers. Three of the models did not attempt to make predictions for specific species of berries but instead provided predictions on a generic basis. In the case of the FINNFOOD model, which had not been used to provide predictions for any of the previous endpoints, two particular species of berry-bearing plant were modelled – bilberry (*Vaccinium myrtillus*) and raspberry (*Rubus idaeus*).

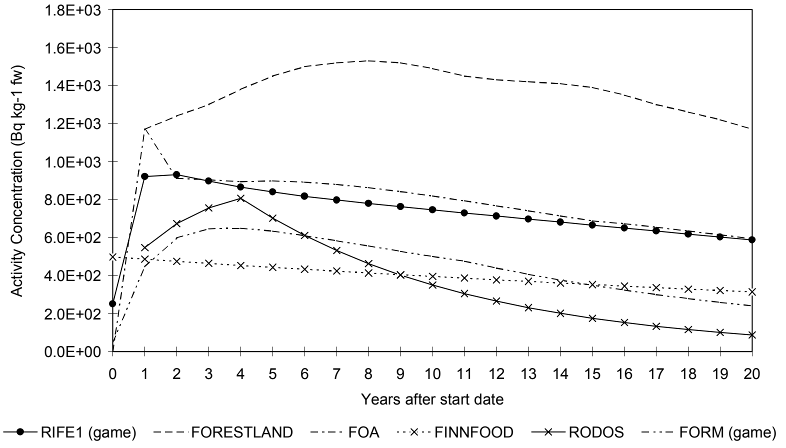


FIG. 14. ^{137}Cs activity concentration in moose (Bq kg^{-1} fresh weight).

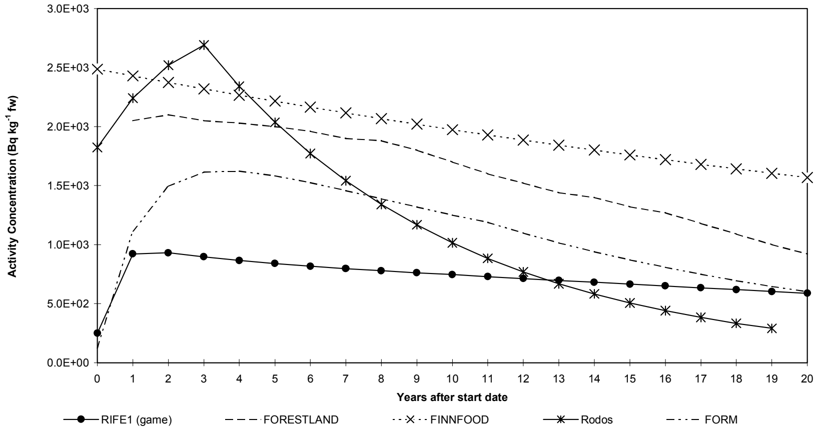


FIG. 15. ^{137}Cs activity concentration in roe deer (Bq kg^{-1} fresh weight).

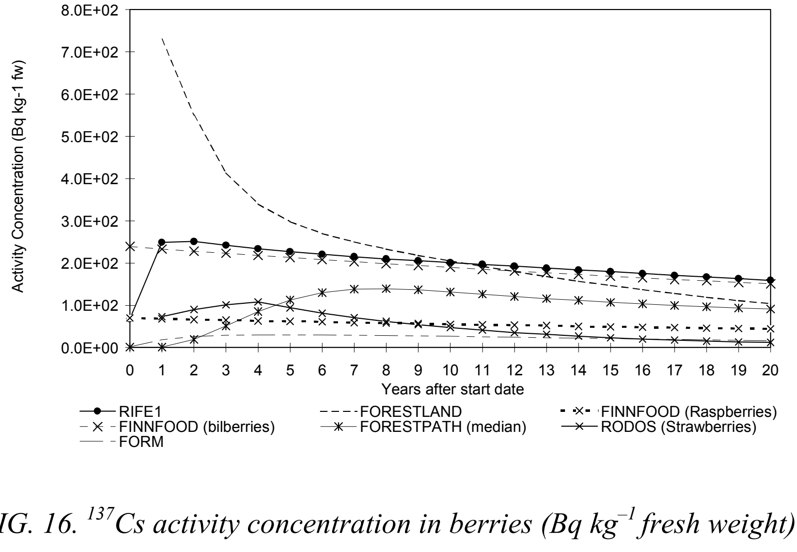


FIG. 16. ^{137}Cs activity concentration in berries (Bq kg^{-1} fresh weight).

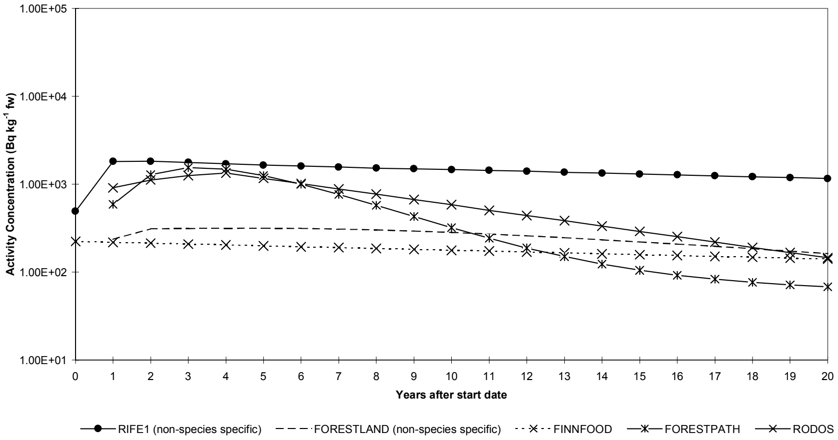


FIG. 17. ^{137}Cs activity concentration in mushrooms (Boletus, Bq kg^{-1} fresh weight).

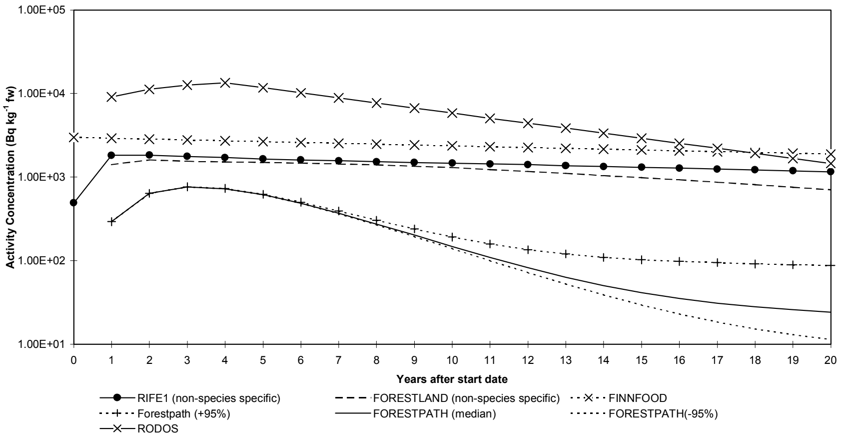


FIG. 18. ^{137}Cs activity concentration in mushrooms (Russula, Bq kg^{-1} fresh weight).

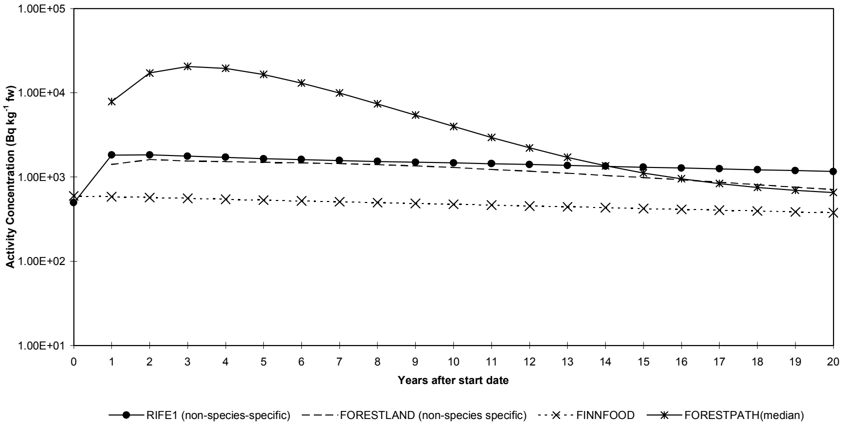


FIG. 19. ^{137}Cs activity concentration in mushrooms (Cantharellus, Bq kg^{-1} fresh weight).

As for the tree predictions, the degree of qualitative agreement between all the predicted time courses was good after the first few years of the simulation. The greatest divergence in predictions during these first five years was provided by FORESTLAND which, as for needles of trees, predicted that an initially maximal degree of contamination immediately after deposition would be followed by a slow, approximately exponential, decline in activity concentration over the 20-year simulation period. This is most probably due to an assumption of considerable external contamination of plant tissues immediately after deposition. None of the other models made this assumption and consequently predicted only a slow decline in activity concentration of berries following a (sometimes rapid) increase in activity concentration of berries. After five to seven years the rate of decline in activity concentrations of berries predicted by each model was very similar and probably reflects the physical decay of ^{137}Cs. The maximum range of predicted activity concentrations during the initial phase following deposition was approximately three orders of magnitude, but less than one order of magnitude during the 10–20 year period of the simulation.

5.4.3.3. Fungal fruiting bodies ('Mushrooms')

One of the confounding factors in the provision of results for Scenario 1 was that different modellers provided results for different mushroom species and it is well known that major species differences occur in terms of ^{137}Cs uptake by mushrooms. In an attempt to clarify the range of predictions received, the mushroom predictions were plotted species-by-species (Figures 17–19). Nevertheless, the variability of predictions for contamination was high, with a range of results spanning about two orders of magnitude for each mushroom species over the entire simulation period (Figures 17–19).

Of the four models from which predictions of mushroom contamination time courses are available most predict qualitatively similar curves over the 20-year period. On closer inspection, however, the models can be divided into different groups with respect to the mushroom contamination dynamics that they predict. FINNFOOD predicts that, having reached peak contamination rapidly (immediately in the case of FINNFOOD) after ^{137}Cs deposition, a steady reduction in mushroom contamination occurs at the same rate as the physical decay of ^{137}Cs. This implies that removal by leaching of ^{137}Cs from the region of the soil exploited by fungal mycelia does not occur during the 20-year simulation period. On the other hand, both the FORESTPATH and RIFE models predict that reduction of mushroom contamination after deposition will occur more rapidly than the physical decay of ^{137}Cs, thereby implying that leaching of radiocaesium from the exploited soil will occur over a 20 year period. In the case of FORESTLAND there is a weak decrease of radiocaesium levels in mushrooms with time because the radiocaesium inventory in available form does not decrease substantially with time. This is despite the fact that a decrease in total radiocaesium levels in the soil exploited by fungal mycelia does occur.

The high degree of variability between the predictions confirms that, conceptually, the modelling of mushroom contamination remains probably the most contentious aspect of forest modelling. This is an important problem both to recognise and solve since, as far as the ingestion dose pathway for forest food products is concerned, consumption of contaminated mushroom is probably the most important single component.

5.5. GENERAL SUMMARY

In brief, the results obtained for Scenario 1 and reviewed in this report have shown that, for the soil and tree compartments of forests, there is generally a high level of consistency between predictions made by the models tested. This is particularly true for the soil compartment and it doubtless reflects the research efforts that have been made over the last 10 years in understanding processes controlling radiocaesium migration in soils.

Predictions of all the biological endpoints proved more variable, especially predictions for mushroom contamination. This raises the question of whether our current (deterministic) modelling approaches are suitable to predict the behaviour of biological entities that will inevitably exhibit a high degree of variability. One or two specific mushroom species will be selected for modelling in future scenarios and this may allow a better understanding of any shortcomings of current modelling approaches (although the Scenario 1 exercise has already demonstrated that predictions for individual mushroom species are currently highly variable).

In addition to these general observations the following summary points were compiled during a meeting of the Forest Working Group to discuss the results of Scenario 1:

- Accurate model descriptions are needed which give accurate definitions of compartments (and interpretation of endpoints) as well as a description of solution algorithms (see Annex I).
- A description of why a particular model structure was used would be helpful in understanding the individual modeller’s conceptual approach as well as an account of how parameter values were obtained.
- As well as reporting results as activity concentrations it is felt that the Forest Working Group modellers should also report results as percentage distributions (or fractional distributions) of ^{137}Cs within the forest ecosystem after deposition. Results may also be more informative if reported without physical decay.
- T_{agg} values (or other appropriate transfer coefficients) for compartments such as trees should be calculated and reported.
- In the model–data inter-comparison (see Section 6) the variability between model predictions should be compared with the variability between data (this may be facilitated by comparison of variability between predicted T_{agg} values).
- The question has been raised as to whether the available models consider tree age and growth adequately; this may form a specific part of a future inter-comparison scenario.
- Compilation of a table of fresh to dry weight ratios for forest products and components is seen as being necessary and desirable. Considerable difficulty was encountered in the preparation of figures of results from Scenario 1 due to the fact that some modellers provided results on a fresh weight basis while others preferred to use a dry weight basis.
- Finally, during discussions within the Forest Working Group it was established that certain processes are currently not (or very poorly) represented in models. One potentially important process is that of foliar absorption, especially by trees, of which our current understanding is very weak.

5.6. STATISTICAL ANALYSIS OF THE RESULTS OF THE MODEL–MODEL INTER-COMPARISON (SCENARIO 1)

5.6.1. Introduction

The objective of the analysis presented here was to analyse the results of the Scenario 1 exercise using an original method developed for evaluating predictive models. The analysis was designed to address the following questions:

- What is the degree of agreement between model predictions?
- Do the models predict similar time dynamics, and does the agreement between the model predictions decrease (do predictions converge) or increase (do predictions diverge) with time?
- For which endpoint (wood, bark, needles, etc.) is the degree of agreement best (or worst).

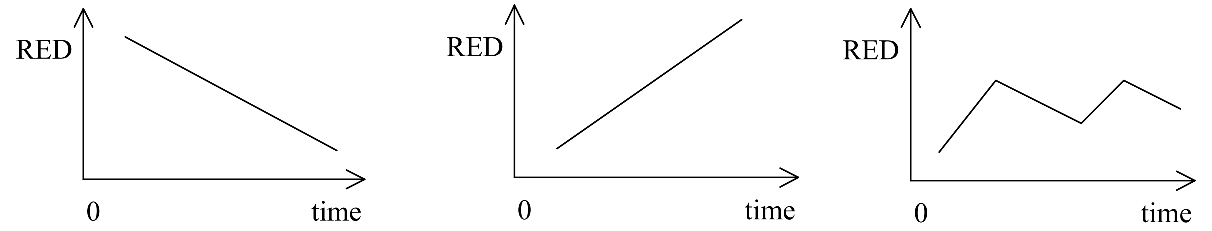
5.6.2. Statistical method of model comparison

The methodology applied here is derived from an original statistical method developed initially to compare predictive models with experimental data (Williams and Leggett, 1983). If only model outputs are compared (with no reference to experimental values) the Relative Euclidean Difference (RED), which is an expression of the sum of differences between every pair of model outputs, is an appropriate tool to evaluate different compartments of a forest ecosystem:

$$RED = \sqrt{\sum_i \left(\frac{x_i - y_i}{x_i + y_i} \right)^2}$$

x_i and y_i refer to the predictions made by each pair of models for the same time "i". The sum contains as many members as the number of possible pair combinations between the "n" models compared (C_n^2).

The RED is a qualitative measure of the average agreement between model predictions. Decreasing (respectively increasing) time series of RED values mean that the predictions of the different models globally converge (respectively diverge) with time. If the RED curve is non-monotonous, the inflexion points correspond to changes in the modelling dynamic of some of the models. If all the models predict the same value (perfect agreement), the RED equals zero (Figure 20).



The models converge with time

The models diverge with time

The models are based on different time dynamics

FIG. 20. Schematic shapes of the evolution of RED with time.

As the RED is generally not constant with time, its maximum value (max(RED), worst case) in a given time series of model predictions has been used to define the reliability index “k” ($0 < k < 1$):

$$k = \frac{1 - \max(RED)}{1 + \max(RED)}$$

5.6.3. Model clustering

The RED analysis highlights the forest compartments for which no clear consensus exists between model predictions. For these compartments, it is interesting to go a step forward and refine the analysis through model clustering. For a given model *m*, this analysis consists in comparing the differences between all the pairs of models (*global* value) with the differences calculated without considering the pairs of models in which the model *m* appears (*limited* value). As clustering is carried out independently for each compartment, the absolute differences between models predictions (Absolute Euclidean Difference – AED) is generally used:

$$AED = \sqrt{\sum_i (x_i - y_i)^2}$$

An individual AED value which significantly diverges from the global AED value means that model *m* differs significantly from the other models for the compartment studied.

5.6.4. Results and discussion

Some models involved in the Scenario1 exercise are generic, others are specific to one or several compartments. Table 4 summarises the ecosystem compartments which are respectively considered by the different models.

The RED values have been calculated for the different compartments according to Table 4. The results, grouped in a logical way (tree, soil, understorey), are presented in Figures 21–23. In these figures, the real value of RED has no clear significance, only the comparison of the relative position of the curves and of their tendencies is meaningful.

Two periods can clearly be distinguished regarding the tendencies of the curves. During the first 5 years after contamination, all models converge for the tree (including litter) and understorey compartments, but diverge for the soil compartments. After this period, the estimates for all compartments diverge or show no clear tendencies.

TABLE 4. MODEL–MODEL COMPARISON: SYNTHESIS

	Total tree	Wood	Needles	Bark	Litter	Organic soil	Mineral soil	Berries	Moose	Roe deer	Mush-rooms
ECORAD-C	×	×	×	×							
FORESTLAND		×	×	×		×	×	×	×	×	×
FOA	×	×	×		×	×	×		×		
FORESTPATH	×					×	×	×			×
RODOS								×	×	×	
RIFE1	×	×	×	×	×	×	×	×	×		×
FORM								×		×	
LOGNAT		×	×		×	×	×				
FORESTLIFE		×	×	×	×	×					
FINNFOOD								×	×	×	×

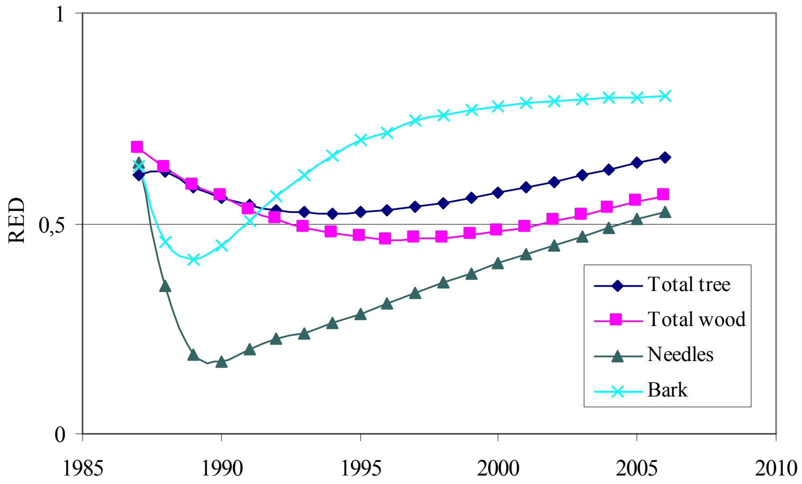


FIG. 21. Time variation of the RED for tree compartments.

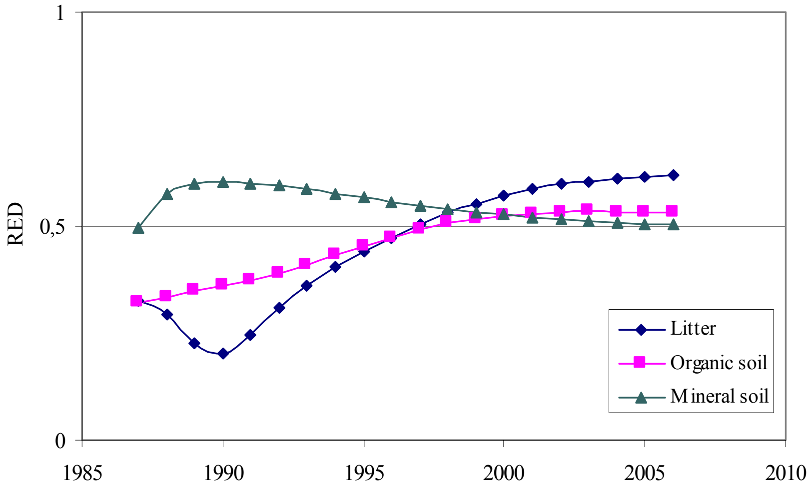


FIG. 22. Time variation of the RED for the litter–soil compartments.

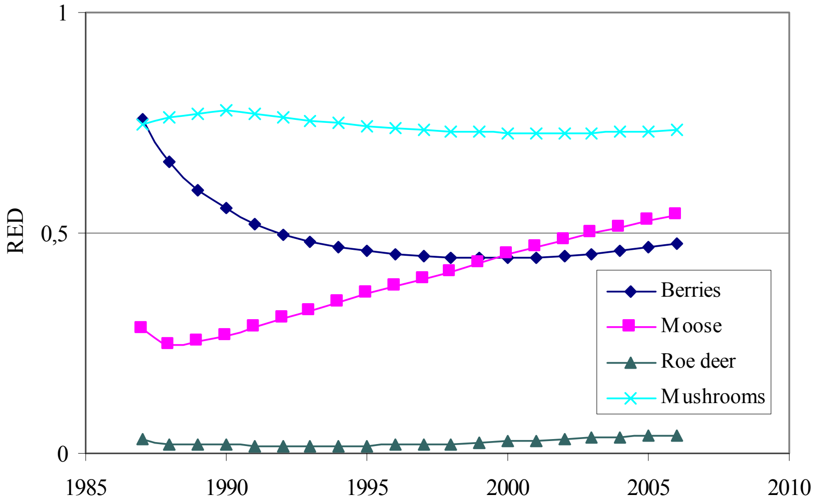


FIG. 23. Time variation of the RED for the understorey compartments.

Values of the reliability index *k* are summarised in Table 5. The different compartments can be classified as follows (from best to worst) : roe deer >> moose, organic soil, mineral soil > total tree, wood, needles, litter >> bark, berries, mushrooms.

TABLE 5. MODEL–MODEL COMPARISON: RELIABILITY INDEX

	Total tree	Wood	Needles	Bark	Litter	Organic soil	Mineral soil	Berries	Moose	Roe deer	Mush-rooms
Agreement (k factor)	0.21	0.19	0.22	0.11	0.24	0.30	0.25	0.14	0.30	0.92	0.13

The rather poor results for berries and mushrooms are probably due to the fact that several different species are considered in the comparison. On the other hand, the very good results for roe deer are likely due to the limited data sets used for calibration of the models.

Model clustering was carried out for needles, bark, wood and litter compartments. The results are presented in Figures 24–27. The model clustering for needles, bark, wood and litter is summarised in Table 6.

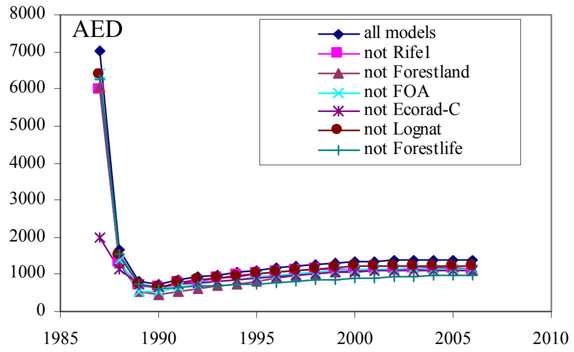


FIG. 24. Cluster analysis (needles).

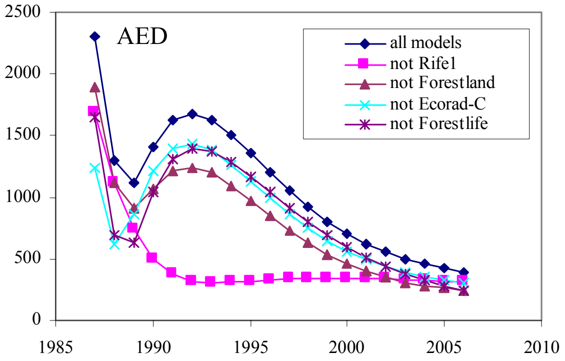


FIG. 25. Cluster analysis (bark).

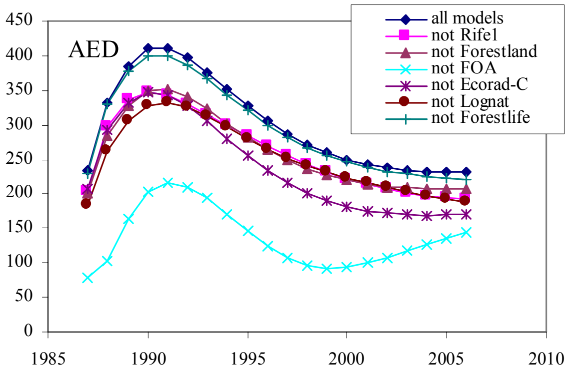


FIG. 26. Cluster analysis (wood).

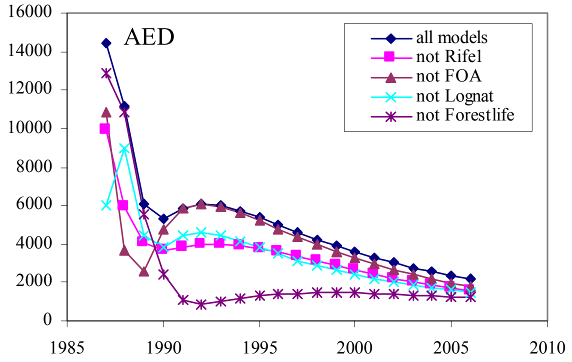


FIG. 27. Cluster analysis (litter).

TABLE 6. MODEL–MODEL COMPARISON: CLUSTER ANALYSIS

Compartment	Model clustering and comments
Needles	2 clusters: 1) ECORAD-C 2) other models ECORAD-C significantly differs just after the contamination (the level in needles is much higher than in the other models)
Bark	2 clusters: 1) RIFE1 2) other models RIFE significantly differs from year 3 (the level in bark increases during 3 years then decreases)
Wood	2 clusters: 1) FOA 2) other models The time dynamics are similar for all the models
Litter	4 clusters: 1) FOA 2) LOGNAT 3) FORESTLIFE 4) other models Only FOA shows no litter increase between year 3 and 6 In LOGNAT, the level in litter decreases between year 1 and 2 FORESTLIFE significantly differs from year 3

The analysis of Figures 24–27 reveals that for needles, bark and litter, a rapid convergence can be obtained by removing only one model. Long-term management strategies can thus be based on a good consensus between models. On the other hand, for wood, the initial phase of divergence of the simulations cannot be attributed to one given model; although the absolute variations are reduced if FOA is not taken into account.

5.6.5. Conclusions of statistical analysis

Radioecological models simulating radiocaesium cycling in forest ecosystems have been compared for the same reference contamination scenario but with no reference to real experimental data. As the different models involved are generic or specific to some forest compartment(s) (tree, soil, understorey), simple statistical tools based on the comparison of models were used, firstly, to identify the tendencies (convergence or divergence) between time series of model estimations and, secondly, for the most varying compartments, to discriminate between models through a clustering analysis.

The agreement between models (k factor) varied between 10 and 30%. The highest agreement was shown by roe deer (92%) but this value is not significant. Globally, all model estimations converge in the first five years following the contamination and show varying tendencies afterwards. Finally, the cluster analysis proved to be a powerful tool to reduce the noise of the global AED curve by isolating the models which are based on different time dynamics.

Considering the high complexity and variability of the phenomena studied, the present models for ^{137}Cs behaviour in forest compartments are in satisfactory agreement. Nevertheless, differences in estimations of time dynamics between models show that a better understanding of the ^{137}Cs behaviour and cycling in forest ecosystems is still needed, especially for wood. Moreover, a comprehensive evaluation of these models would ideally require a comparison of model outputs with experimental data to make the link with long-term management strategies of real cases of forest ecosystem contamination.

6. MODEL–DATA INTER-COMPARISON STUDY – ‘SCENARIO 2’

6.1. INTRODUCTION

This section describes results from the second model inter-comparison study undertaken by the Forest WG. The study involved comparison of outputs from 9 different models (see Section 6.3). The contamination scenario (‘Scenario 2’) that was posed for this exercise was based on a data set obtained for a forest site near Kiev, Ukraine, contaminated with ^{137}Cs in 1986 following the Chernobyl accident. Scenario 1 (Section 5) was hypothetical and was intended not to validate model predictions against actual data but to compare the results produced by each modeller and his/her respective model. The primary objective of Scenario 2, however, was to allow a direct comparison of model predictions against a time series of data collected at a forest site which non of the modellers had previously seen and for which they had no prior information. In this sense the exercise was conducted ‘blind’ and the outcome of the exercise was not revealed to the modellers until the results obtained from individual models had been collated and plotted against the actual time series data for the forest site in question. Some problems of interpretation of results did arise due to the fact that not all of the models were constructed to make predictions of each of the endpoints specified in the scenario. Therefore, some work was required after the results of the exercise had initially been revealed to the Forest WG. In some cases, second round results were submitted, although these are not dealt with in this report (the models concerned were the FORSUN and FORESTPATH models; further information can be obtained from the authors of these models, as listed in Annex I). Each of these stages in the model–data inter-comparison is described below and a description of results is provided.

6.2. SUMMARY OF SCENARIO 2

The complete scenario is listed in Annex III. The scenario concerned a forest situated near Rudnya-Povcha in the Zhitomir region of Ukraine, approximately 130 km to the south-west of the Chernobyl power plant (51° 09’ N, 28° 35’ E). For convenience, the date of deposition of ^{137}Cs was taken as 1st May 1986: the total deposition at this time was 555 kBq m^{-2}. The main soil type at the site is a soddy-podzolic sandy loam which is characterised by low natural fertility and high permeability to water flow. The dominant tree species is Scots pine (*Pinus sylvestris*) with an average age of 50 years, accompanied by sparse birch (*Betula pubescens*) with an average age of 25 to 30 years. The understorey is dense and comprises several species of bilberry (*Vaccinium* spp.), purple moor grass (*Molinia caerulea*) and various other herbaceous species of lesser importance. Mushroom species to be found at the site include *Boletus edulis*, *Suillus luteus*, *Cantharellus cibarius*, *Xerocomus badius* and *Russula paludosa*. The main game species at the site is roe deer.

Participants were requested to predict activity concentrations, on a dry weight basis, of the following endpoints:

- wood of *Pinus sylvestris*;
- annual shoots of *Pinus sylvestris*;
- needles of *Pinus sylvestris*;
- total bark of *Pinus sylvestris*;
- soil profile, including litter;
- roe deer;
- ‘mushrooms’ (individual species to be modelled at the discretion of each modeller);
- bilberry.

As mentioned in the introduction, not all of the models were designed to predict radiocaesium concentrations in all of the required endpoints. For this reason, participating modellers were given the freedom to report on their own choice of endpoints selected from the above.

It was requested that each chosen endpoint should be considered as a function of time over the period 1986 to 1998, with results to be reported at one-year intervals. It was further requested that calculations be based on best estimates of parameter input values and that, if possible, results be reported as both 'best estimates' and/or 95% confidence intervals: in fact, only one model (FORESTPATH) produced both best estimates and 95% uncertainty ranges.

6.3. PARTICIPANTS AND MODELS

Individual modellers and their respective models are listed in Table 1. Brief descriptions of each of these models are provided in Annex I.

6.4. RESULTS

The results are presented graphically as 12-year time courses (1986–1998) following initial deposition. As requested in the scenario sent out to respondents, the resolution of these predicted time courses is one year.

All the modellers reported results as 'best estimates'. The results obtained with the FORESTPATH model were also reported as 95% confidence intervals. To simplify the graphs, normally only the median value calculated with the FORESTPATH model is plotted. The 95% confidence intervals are only plotted when they differ significantly from the median value.

6.4.1. Trees and associated components

Predictions of activity concentrations of specific components of trees (*Pinus sylvestris*) were provided by six respondents, although only the FORESTLAND and FOA models returned results for each of the endpoints requested. Figures 28–31 show plotted results for each of the specific tree-related endpoints, namely wood (without bark), total bark (including cambium), annual shoots and needles.

The first impression from Figures 28–31 is that the range of model predictions for each of the tree-related endpoints generally encompasses the measured data at each time interval. This observation applies particularly to predictions for wood and total bark (Figures 28 and 29) in which, at all time points, the measured data values fall within the range of values predicted by each of the models concerned. This immediately suggests that, for the Zhitomir site, the predictions provided by the FORESTLAND, RIFE, ECORAD, FOA, FORM and S-RODOS models could be used collectively to provide a reliable envelope of model predictions within which measured data on wood and bark activity concentration would be expected to fall. For predictions of annual shoots and needles (Figures 30 and 31), however, a significant number of measured data fell outside the envelope of model predictions, although in the case of annual shoots predictions were only provided by three respondents. This comparison demonstrates the value of multiple model predictions applied to a single data set: despite differences in the predictions of individual models there are evidently no 'wild card' predictions and it is possible that, using the models employed in this study, an averaged or 'consensus' prediction could be obtained which would agree quite well with the measured data.

Results for each of the individual tree-related endpoints will now be examined in turn.

Of the five predictions that were made of the time-course of contamination of wood (Figure 28) four indicated very similar dynamics. These were FORESTLAND, FOA, RIFE and FORM, which all showed an initial increase in activity concentration from very low (effectively zero) initial values. FORESTLAND, FOA and RIFE provided predictions that were particularly close to the measured data. FORM predicted very similar dynamics to these three models, but predicted significantly lower activities in wood for most of the 12-year period. ECORAD predicted an exponentially declining activity in wood from the very beginning of the 12-year period, suggesting a significantly different conceptual approach.

For total bark (Figure 29) five models provided predictions. Of these, FORESTLAND, ECORAD, S-RODOS and RIFE all provided predictions which closely bracketed the measured data over the period 1991 to 1998, although the dynamics of these models over the 12 year assessment period varied considerably. S-RODOS and RIFE both indicated an initial sharply declining activity in bark, suggesting an initially important component of external contamination immediately after deposition of ^{137}Cs from the atmosphere. ECORAD and FORESTLAND both indicated an exponentially declining trend over the whole assessment period, although without the initially very high external activities predicted by FINNFOOD and RIFE. FOA consistently predicted higher and more-or-less constant bark activities than the other four models.

Only three modellers provided predictions for annual shoot activities (Figure 30). These were FORESTLAND, FOA and S-RODOS. The scatter in the measured data was considerable for this endpoint, and each of these models provided predictions which lay more-or-less centrally within this scatter of data, even though each model predicted different ‘early’ dynamics immediately after deposition.

Six modellers provided predictions for needle activities (Figure 31). The models concerned were FORESTLAND, ECORAD, RIFE, FOA, S-RODOS and FORM. During the period 1991 to 1998, for which measured data were available, the dynamics and range of predicted values of each of the models was remarkably consistent, with less than one order of magnitude spanning the highest (FORESTLAND) and lowest (FORM/ECORAD) predictions. 50% of the measured data points were within the range of predicted values provided by the models.

It should be noted that, while the agreement between model predictions and data was impressive for the period 1991 to 1998, for which data are available, there are large discrepancies in model predictions for the ‘early phase’, approximately 1986 to 1988, for which no data are available from the Zhitomir site.

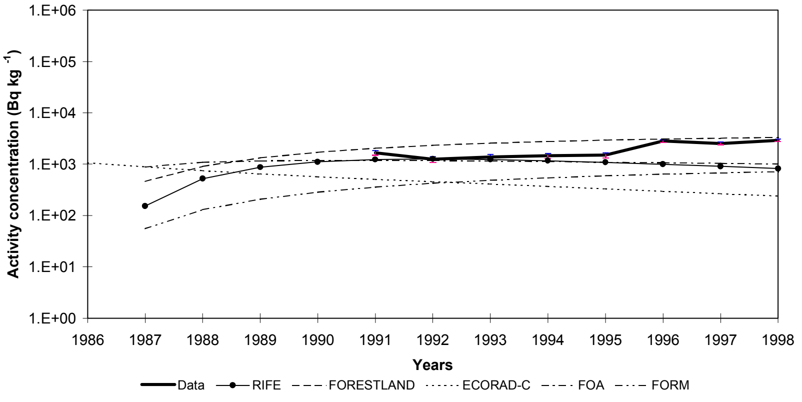


FIG. 28. ^{137}Cs activity concentration in wood (without bark, Bq kg^{-1}).

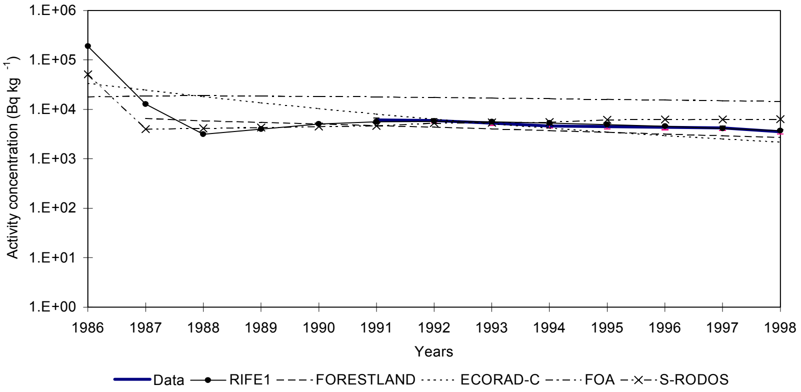


FIG. 29. ^{137}Cs activity concentration in bark (including cambium) (Bq kg^{-1}).

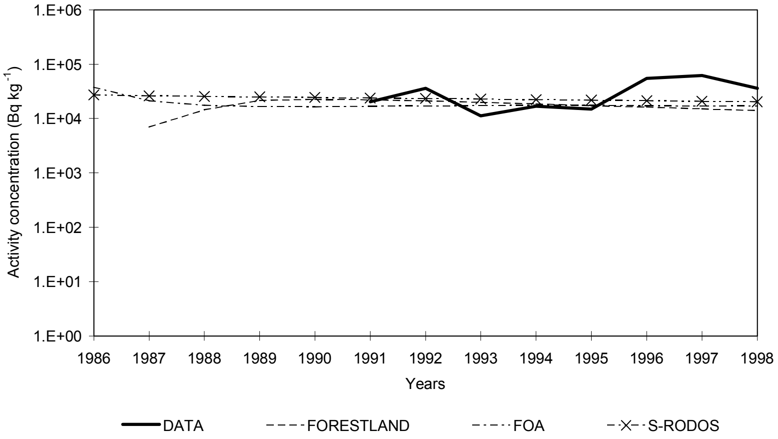


FIG. 30. ^{137}Cs activity concentration in annual shoots (Bq kg^{-1}).

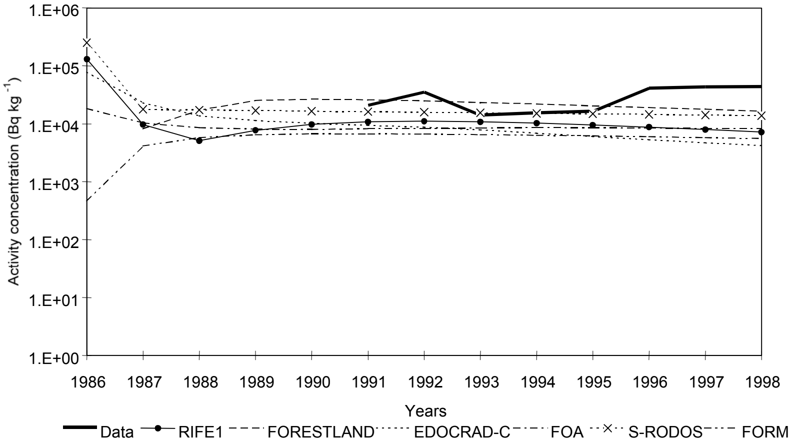


FIG. 31. ^{137}Cs activity concentration in needles (Bq kg^{-1}).

6.4.2. Soils

Model predictions for soil contamination over the 12 year assessment period were submitted for five models, FORESTLAND, FORESTPATH, FOA, FORSUN and RIFE. Direct comparison of all models for all soil endpoints was made impossible by the fact that some models are designed to return averaged predictions for some soil layers. Data from the Zhitomir site were available for the following soil layers:

—	AoL	litter;
—	AoF	fermentation horizon;
—	AoH	humified horizon;
—	Ah	dark, mineral-humic horizon;
—	E	bleached, eluvial horizon.

Each of the models provided either horizon-specific results or results which are averaged over more than one horizon.

Figure 32 shows predictions of RIFE, FOA, FORESTPATH and FORSUN for the AoL layer. Both RIFE and FOA overpredict the measured data by approximately one order of magnitude, whereas FORESTPATH and FORSUN both make more accurate predictions of litter activities, especially over the period 1993 to 1998. The dynamics of litter activity predicted by FORSUN, FORESTPATH and RIFE are all quite similar: each model predicts a decline in litter activity after the first year following contamination. FOA predicts a continuing build-up of activity with a plateau establishing after approximately 5 years.

Figure 33 shows predictions of FOA, FORESTPATH and FORSUN of activity in the AoF layer. Again, FORESTPATH and FORSUN show similar dynamics and are both quite accurate in their predictions. FOA shows rather different dynamics and tends to under predict the measured data, possibly as a result of over predicting the litter activities.

Figure 34 shows predictions of FOA, FORESTPATH, FORESTLAND and FORSUN of activity in the AoH horizon. FORESTPATH, FORESTLAND and FORSUN each show very similar dynamics and accuracy in reproducing the measured data. FOA again under predicts the measured data by approximately one order of magnitude.

Data for the 10-cm thick Ah horizon were presented for individual 2cm thick slices (5 in all). Only FORESTLAND and FORSUN provide predictions for these individual 2 cm slices and, as an illustration, the result of FORESTLAND and FORSUN predictions for the uppermost 2 cm slice of the Ah horizon is shown in Figure 35. As for previous soil endpoints, both of these models performed extremely consistently and accurately with both predicting both the magnitude and time course of Ah activity very accurately.

The RIFE and FOA models provided averaged predictions for the Ah horizon, shown in Figure 36. Both models predicted the dynamic accumulation of activity within the averaged Ah horizon, although the FOA model under predicted the Ah activities by slightly less than one order of magnitude.

The conclusions from this part of the exercise show that some extremely accurate predictions of soil contamination are possible with the models put forward in this study. One major problem in a model inter-comparison such as this, however, is that different model structures do not always allow a direct comparison between predictions. This is particularly evident in the case of soils in which conceptual subdivisions of the soil column are artificial and reflect (a) the individual requirements of a model; and (b) the individual conceptual approach of the modeller concerned.

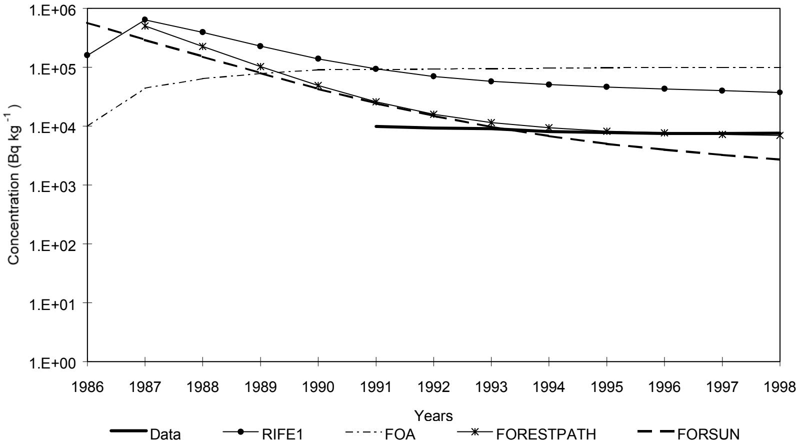


FIG. 32. ^{137}Cs activity concentration in soil profile (layer AoL) (Bq kg^{-1} dry weight).

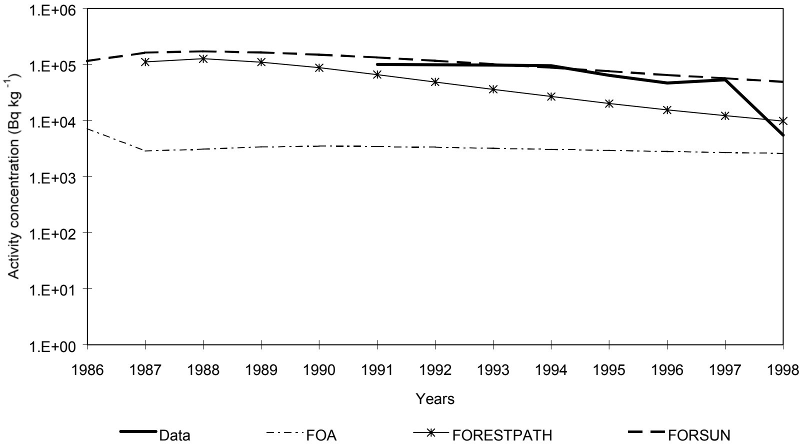


FIG. 33. ^{137}Cs activity concentration in soil profile (layer AoF) (Bq kg^{-1} dry weight).

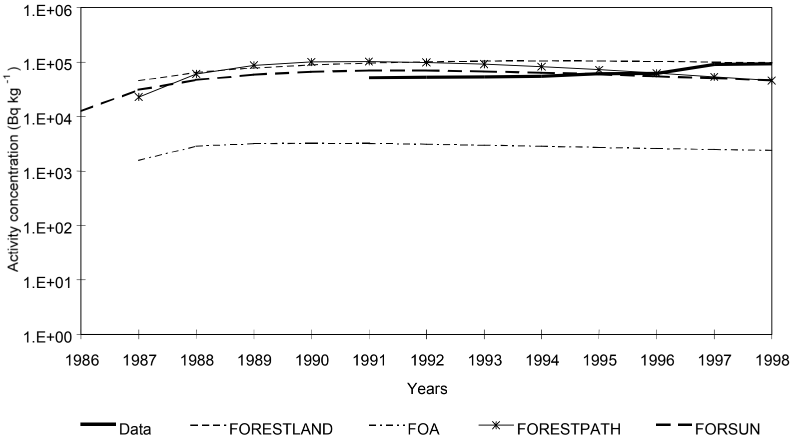


FIG. 34. ^{137}Cs activity concentration in soil profile (layer AoH) (Bq kg^{-1} dry weight).

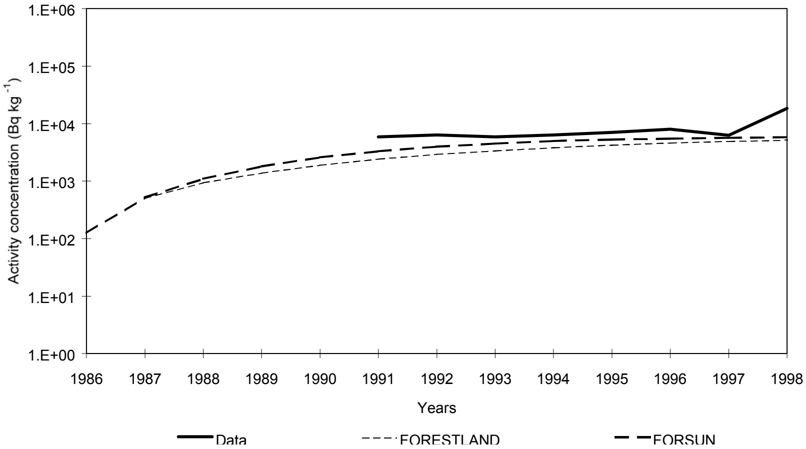


FIG. 35. ^{137}Cs activity concentration in soil profile (first slice of Layer Ah) (Bq kg^{-1} dry weight).

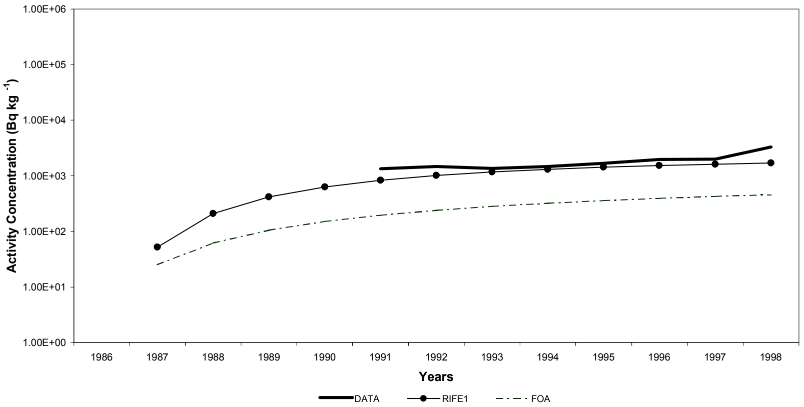


FIG. 36. Averaged ^{137}Cs activity concentration in soil profile (total layer Ah) (Bq kg^{-1} dry weight).

6.4.3. Other biological endpoints

The biological endpoints of concern in Scenario 2 are roe deer, bilberries and mushrooms (various species). Model predictions of contamination time courses for each of these components of the forest ecosystem are shown in Figures 37–39 respectively.

6.4.3.1. Roe deer

Five modellers provided predictions for time courses of contamination in roe deer. Of the five models, two (FORM and RIFE) interpret game in a generic sense while the remaining three specify the species of interest. The spread of predictions produced by these models (Figure 37) is relatively large, spanning two to three orders of magnitude for the period for which measured data are available. Both FOA and FORESTLAND produced accurate predictions of ^{137}Cs activity in roe deer, both with similar dynamics. The other three models (FORM, RIFE and RODOS) produced predictions which were consistent with each other but which were approximately one to two orders of magnitude below the measured values. Only RODOS predicted a significantly higher activity in roe deer immediately following contamination in 1986, but no measured data are available to verify this prediction.

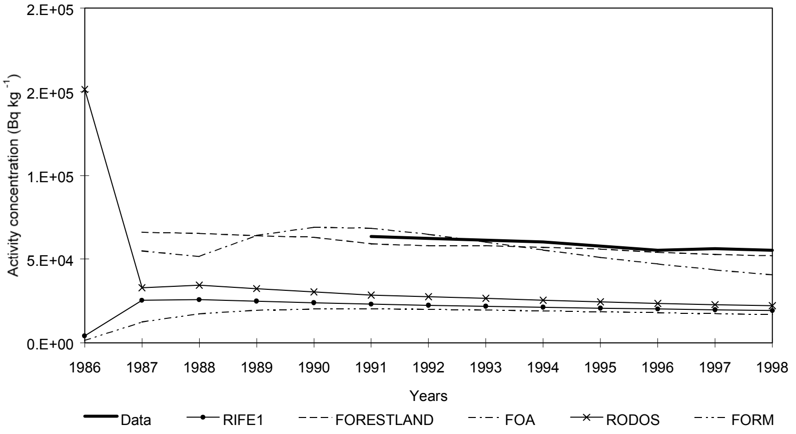


FIG. 37. ^{137}Cs activity concentration in roe deer (Bq kg^{-1} dry weight).

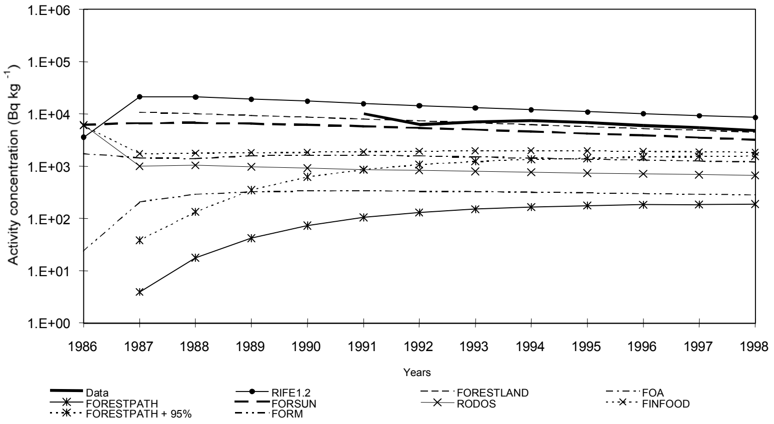


FIG. 38. ^{137}Cs activity concentration in bilberries (Bq kg^{-1}fresh weight).

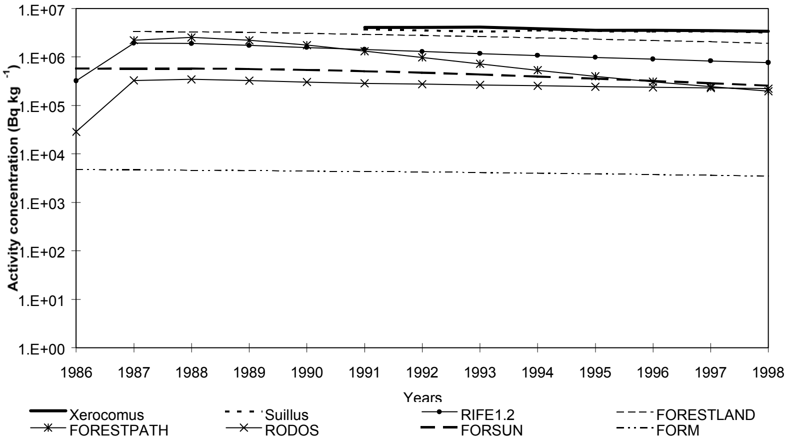


FIG. 39. ^{137}Cs activity concentration in mushrooms, Xerocomus badius and Suillus luteus (Bq kg^{-1}dry weight).

6.4.3.2. Bilberry

Predictions of time courses of contamination of the understorey focussed on bilberries since the dominant herbaceous species present at the Zhitomir site are *Vaccinium* species. Results were provided by eight of the modellers. The range of predictions was large, spanning 3 to 4 orders of magnitude (Figure 38). This range of predicted values was asymmetrically distributed around the measured values, with most of the models under predicting the measured data significantly. RIFE, FORESTLAND and FORSUN each made predictions significantly less than one order of magnitude different from the measured data. It is noticeable that the dynamics of bilberry contamination predicted by each model are very similar for the 1991–1998 period for which data are available, suggesting that most modellers agree on the processes that contribute to ^{137}Cs contamination of bilberry, and that discrepancies between models are most likely due to differences in calibration.

6.4.3.3. Mushrooms

Results were returned for *Xerocomus badius* and *Suillus luteus* (Figure 39), *Russula paludosa* and *Boletus edulis* (Figure 40) and *Cantharellus cibarius* (Figure 41) using each of the models with the exception of ECORAD. For brevity all of these results can be described as an ensemble, since the predictions for each of the mushroom species were very similar. Each of the predictions was characterised by a) a broad spread of predicted values (generally three to four orders of magnitude) and b) a generally significant under prediction of the measured values by each of the models. Individual models did perform well for single species (for instance FORESTLAND performed well for *Xerocomus badius* and *Suillus luteus* and FORESTPATH performed well for *Cantharellus cibarius*) but in general the performance of the models was not as good as it was for other end points.

It was concluded in Scenario 1 (see Section 5) that a high degree of variability between the predictions for mushrooms indicates that the modelling of mushroom contamination remains the most problematic aspect of forest modelling. The results of the blind predictions in Scenario 2 confirm this view.

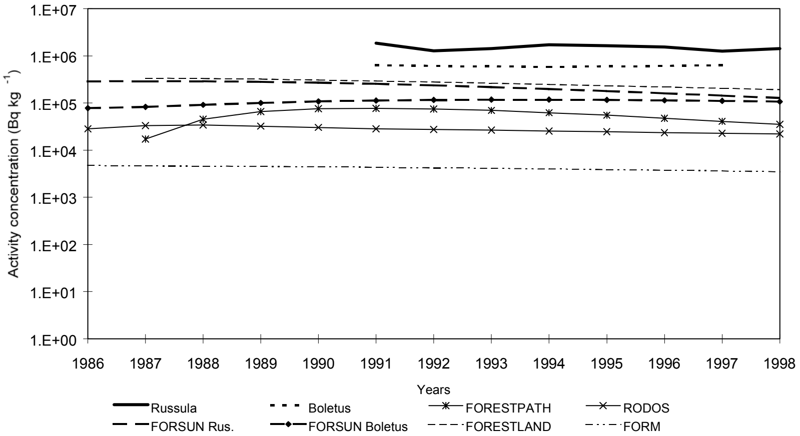


FIG. 40. ^{137}Cs activity concentration in mushrooms, Russula paludosa and Boletus edulis (Bq kg^{-1}dry weight).

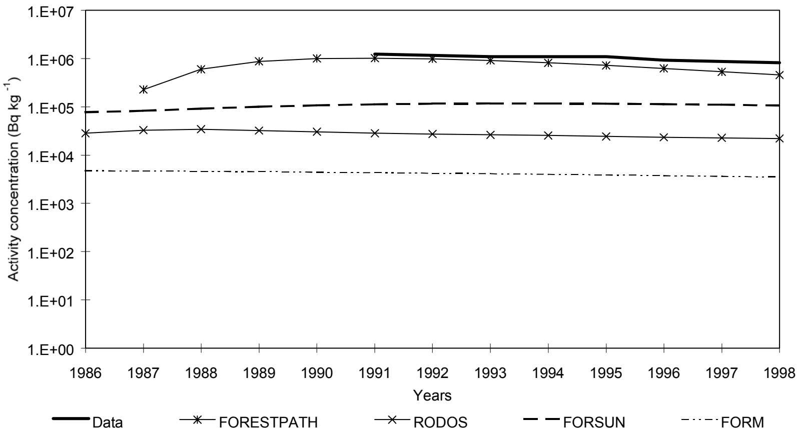


FIG. 41. ^{137}Cs activity concentration in mushrooms, Cantharellus cibarius (Bq kg^{-1} dry weight).

6.5. GENERAL SUMMARY

The results obtained for Scenario 2 in general support the conclusions drawn from Scenario 1. Hence, for many of the conceptual compartments represented within the forest models available to the Forest WG there is generally a high level of consistency between predictions made by the models tested. Furthermore, when compared with the Rudnya-Povcha data set, many of the model predictions proved rather accurate, although admittedly over a period some 5 to 12 years after initial contamination when the 'early' dynamics of ^{137}Cs within the forest ecosystem are likely to have been superseded by slower, long-term rates of redistribution within the forest. Particularly accurate and consistent predictions were made for the tree-related compartments and for certain soil compartments.

Yet again, however, the biological endpoints (roe deer, bilberries and mushrooms) proved more difficult to model. Predictions of all these endpoints proved more variable and, sometimes, consistently inaccurate (especially predictions for mushroom contamination).

6.6. STATISTICAL ANALYSIS

6.6.1. Objective

The objective of the analysis presented here was to apply the statistical methods described in Section 5.6.2 to the results of the model–data inter-comparison. The specific questions addressed in this analysis were as follows:

- What is the degree of agreement between model predictions and experimental data?
- For which endpoint (wood, bark, needles) is the degree of agreement the best or the worst?

6.6.2. Statistical method

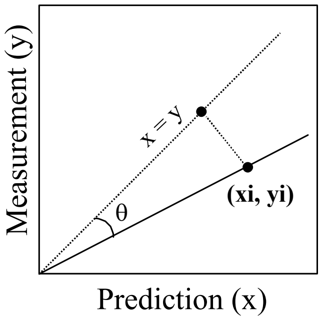
The methodology applied is derived from an original statistical method proposed by Williams and Leggett [1984] for comparison of predictive models against experimental data. When comparing model predictions with experimental data, two distinct types of uncertainties have to be considered:

(a) uncertainty associated solely with the model, which usually arises from incomplete understanding of the phenomenon being modelled; and

(b) uncertainty associated with the observations, arising from the inherent variability of the phenomenon being measured and from imprecision in the measurement procedure.

According to Williams and Leggett [1984], two reliability indices can be used as a measure of the accuracy of a model: a geometrically intuitive reliability index, k_g, and a statistically rigorous reliability index, k_s. For reasonably accurate models ($k<2$), k_g and k_s can be used interchangeably as a reliability index (it was shown that, in these conditions, $0.989 < k_g/k_s < 1.027$), but the geometrical definition of the reliability index is easier to conceptualise. If there is a time set $\{y_1, ..., y_n\}$ of observations corresponding to a time set $\{x_1, ..., x_n\}$ of model predictions, the goodness of the prediction x_i is determined by the relative proximity of the point (x_i, y_i) to the line $x=y$.

The definition of the reliability index k_g is then obtained as follows:



$$\mathrm{Tan}(\theta) = \frac{\sqrt{(x_i - y_i)^2}}{\sqrt{(x_i + y_i)^2}}$$

$$s = \sqrt{\frac{1}{n}\sum_{i=1}^{n} \tan^2 \theta_i} = \sqrt{\frac{1}{n}\sum_{i=1}^{n} \left(\frac{x_i - y_i}{x_i + y_i}\right)^2}$$

$$k_g = \frac{1+s}{1-s}$$

Therefore:

$$k_g = \frac{1 + \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(\frac{1-\frac{y_i}{x_i}}{1+\frac{y_i}{x_i}}\right)}}{1 - \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(\frac{1-\frac{y_i}{x_i}}{1+\frac{y_i}{x_i}}\right)}}$$

6.6.3. Results

The model predictions were compared with yearly means of experimental on-site values measured between 1991 to 1998. In the case of roe deer and mushrooms experimental values were missing for some years.

The results of the comparison are summarised in Tables 7 to 11.

TABLE 7. MODEL–DATA COMPARISON: k_s VALUES FOR TREE COMPARTMENTS

Tree parts					
	Wood	Bark	Needles	Shoots	Average
ECORAD-C	5.75	1.37	4.73	5.08	4.23
FORESTLAND	1.60	1.33	1.83	2.21	1.74
FOA	1.88	3.41	3.40	2.09	2.70
FORESTPATH	5.18	12.74	65.07		27.66
S-RODOS		1.38	2.10	1.91	1.80
RIFE1	2.03	1.08	3.21		2.11
FORM	3.47		4.57		4.02
Average	3.32	3.55	12.13	2.82	
St. Dev	1.50	4.98	25.35	0.15	

TABLE 8. MODEL–DATA COMPARISON: k_s VALUES FOR SOIL COMPARTMENTS

Soil profile							
	Ol (0–2 cm)	Of (2–6 cm)	Ol+Of (0–6 cm)	Oh (6–8 cm)	Ah (8–10 cm)	Ah (average)	Average
FORESTLAND			1.62	1.69	2.07		1.79
FOA	11.70	20.70	2.21	23.16		5.45	12.64
FORESTPATH	1.50	2.78	2.21	1.67			2.04
FORSUN	1.93	2.21	1.68	1.45	1.70		1.79
RIFE1	6.39					1.42	3.91
Average	8.22	10.37	3.50	9.36	1.89	3.43	
St. Dev	5.36	9.27	2.98	10.18	0.26	2.85	

TABLE 9. MODEL–DATA COMPARISON: k_s VALUES FOR MUSHROOMS

Mushrooms						
	Cantharellus	Russula	Boletus	Xerocomus	Suillus	Average
FORESTLAND		6.34	2.52	1.58	1.49	2.98
FORESTPATH	1.47	27.25	10.70	8.38	8.50	11.26
RODOS	41.08	6.06	24.12	14.99	13.87	20.02
FORSUN	9.05	8.25	5.40	10.18	9.67	8.51
RIFE1				9.33		9.33
FORM		385.42	154.43	956.91	883.52	595.07
Average	17.20	86.66	39.43	166.90	183.41	
St. Dev	21.03	167.25	64.82	387.05	391.40	

TABLE 10. MODEL–DATA COMPARISON: k_s VALUES FOR BILBERRIES

Bilberries	
FORESTLAND	1.17
FORESTPATH	43.22
RODOS	13.25
FORSUN	1.54
RIFE1	1.76
S-RODOS	3.53
FORM	21.33
Average	12.09
St. Dev	17.12

TABLE 11. MODEL–DATA COMPARISON: k_s VALUES FOR ROE DEER

Roe deer	
FORESTLAND	1.06
FOA	1.20
RODOS	2.36
RIFE1	2.81
FORM	3.17
Average	2.12
St. Dev	0.95

A summarised estimation of the accuracy of the models for all studied endpoints is presented in Table 12. All tree compartments are presented under “Tree parts”. The soil compartments are subdivided into an organic and a mineral part. In this table, “3”, “2” and “1” indicate that model predictions and the experimental data agree within a factor of 2, 10 and >10, respectively. If the endpoint was not estimated by the model then the corresponding cell remains blank. The last column is obtained by summing the values for all endpoints and can be seen as a global index of the model predictive capacity and accuracy. According to this index, the models can be ranked as follows (from best to worst):

FORESTLAND, RIFE1 > FOA, FORESTPATH, FORSUN,
S-RODOS > ECORAD-C, RODOS, FORM

TABLE 12. MODEL–DATA COMPARISON: SYNTHESIS

	Tree parts	Organic soil	Mineral soil	Mushrooms	Bilberries	Roe deer	Total
ECORAD-C	2						2
FORESTLAND	3	3	3	3	3	3	18
FOA	3	1	2		3	3	12
FORESTPATH	1	3		2	1		7
RODOS				1	1	3	5
FORSUN		3	3	2	3		11
RIFE1	3	2	3	2	3	3	16
S-RODOS	3				3		6
FORM	2			1	1	3	7

6.6.4. Conclusions of statistical analysis

Except for mushrooms, and to a lesser extent for berries, the predictions made by the models were in good agreement with the experimental data (the reliability index was less than 5 in all cases). Overall, FORESTLAND and RIFE1 give the best results (Table 12). Notwithstanding the accuracy of these models, this is probably partly due to the fact that they were calibrated with measurements from the same, or similar, geographical area as the reference scenario. Appropriate calibration is a powerful determinant of model accuracy and it might be expected that the results of the model inter-comparison would have been different if data obtained from another region had been selected as the reference scenario. Finally, the conclusions about model accuracy have to be treated with care because they are based on measurements over a relatively short period (8 years) compared to the time scale of the biological and geochemical processes in a pine forest ecosystem.

This inter-comparison exercise has shown that, in general, there is a satisfactory agreement between the blind predictions of nine existing models and measured data on ^{137}Cs behaviour in multiple forest compartments. Differences in time dynamics estimations between models show, nevertheless, that a better understanding of the ^{137}Cs behaviour and cycling in forest ecosystems is still needed. This requires continued collection of experimental data, which will also assist in improving representation of this behaviour in models.

7. SECOND MODEL–MODEL INTER-COMPARISON STUDY – ‘SCENARIO 3’

7.1. INTRODUCTION

This section describes results from the third model inter-comparison study undertaken by the Forest WG. This study, ‘Scenario 3’, involved a model–model comparison exercise similar to Scenario 1. However, Scenario 3 was radically different from the previous two scenarios in that a subterranean source of ^{137}Cs was considered. One of the conclusions of the Scenario 2 exercise was that the generally high success of each of the models in predicting the dynamics of ^{137}Cs at the Rudnya-Povcha site could be due in part to the fact that all models had been developed and calibrated for a Chernobyl-type source term. Discussion within the Forest WG revealed that very few participants had considered a scenario which, instead of involving a discrete deposition event from the atmosphere, consisted of a prolonged or ‘chronic’ release to the soil from an underground source.

The scenario developed for the third inter-comparison was based on a hypothetical, though realistic, case in which a shallow waste repository containing ^{137}Cs had been capped by a clean 1 m thick cover. The details of the repository, described in Section 7.2, were based on a previous IAEA study on quantitative acceptance criteria for near surface disposal of radioactive waste [IAEA, 1999]. It was assumed that, at the time of capping of the waste-filled trenches, no vegetation existed on the soil surface and, therefore, modellers had to take into account the development of a forest cover over a period of 50–200 years through the process of natural regeneration. A summary of Scenario 3 is provided in the next few paragraphs.

7.2. SUMMARY OF SCENARIO 3

The complete scenario is listed in Annex IV. The source term is a series of ten minimally engineered trenches filled with loosely tipped radioactive waste containing ^{137}Cs. The waste material is covered with a 1 m thick layer of clean soil; though this soil is initially devoid of vegetation, a tree cover becomes naturally established following closure of the repository. It was requested that participants provide predictions of ^{137}Cs activity concentrations in the end points listed below over a period over 50–200 years following repository closure. A maximum simulation period of 200 years takes ^{137}Cs through 6.7 physical half-lives, which still leaves a significant activity within the system. One of the interesting questions to be addressed by this scenario was whether the ^{137}Cs activity concentrations within the soil and biological endpoints achieved steady state over this period.

The dominant tree species assumed was Scots pine (*Pinus sylvestris*) with sparse examples of birch. Two hundred years after closure the average height of the trees was assumed to be 20–25 m and the average density of wood biomass on the trenches between 10 and 20 kg m^{-2}. No information on tree growth rates is available over this period. Of particular importance to this scenario were assumptions concerning tree root distributions within the trench caps, which were as follows:

- pine root growth rate decreases with age;
- pine roots reach half maximum depth after 10–15 years;
- root distribution of pine trees older than 40–60 years do not change significantly with time.

Full details of pine root distribution for 12 year old trees were provided in the scenario, described in Annex IV. The end points for which model predictions of activity concentrations (dry weight) were requested were as follows:

- total tree, *Pinus sylvestris*;
- total wood, *Pinus sylvestris*;
- needles, *Pinus sylvestris*;
- total bark, *Pinus sylvestris*;
- ‘mushrooms’ (specifically *Xerocomus badius* and *Boletus* species);
- bilberry *(Vaccinium macrocarpon)*;
- soil – 10 cm depth increments from 0 to 110 cm from the surface.

As in previous scenarios, not all of the models were designed to predict radiocaesium concentrations in all of the required endpoints. One specific problem was that some models made predictions of activity concentrations for soil depths averaged over relatively larger increments, rather than from specific depth increments.

7.3. PARTICIPANTS AND MODELS

The individual modellers, and their respective models, which participated in Scenario 3 are listed in Table 13. The reduction in the number of participants compared with previous scenarios reflects the fact that this scenario was novel and some of the models which had been designed and written to address a Chernobyl-type scenario could not easily be modified to simulate tree uptake from a subterranean source term.

TABLE 13. MODELLERS AND MODELS PARTICIPATING IN SCENARIO 3 OF THE FOREST WG MODEL INTER-COMPARISON. BRIEF DESCRIPTIONS OF EACH OF THESE MODELS ARE PROVIDED IN ANNEX I

Modeller(s)	Model	Institute
R. Avila and L. Moberg	FORESTLAND	SSI, Stockholm, Sweden
S. Fesenko and S. Spiridonov	FORESTLAND	RIARAE, Russia
R. Bergman	FOA	NDRE, Umea, Sweden
M. Frissel	FORM	IAEA, Vienna, Austria
A. Konoplev and A. Bulgakov	FORWASTE	Typhoon, Obninsk, Russia
I. Linkov	FORESTPATH	Harvard University, USA
S. Mamikhin	ECORAD-C	MSU, Moscow, Russia
G. Shaw	RIFE	Imperial College, UK

7.4. RESULTS

The results are presented graphically (Figures 42–51) as time courses over 50 to 200 years following initial trench closure. All the results are reported as ‘best estimates’ only. For some endpoints some modellers provided results for a 50-year simulation period only.

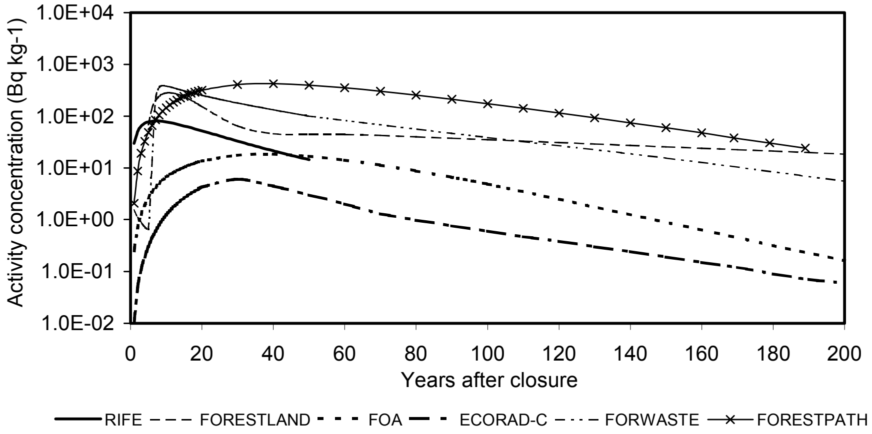


FIG.42. ^{137}Cs activity concentration in total tree (Bq kg^{-1}).

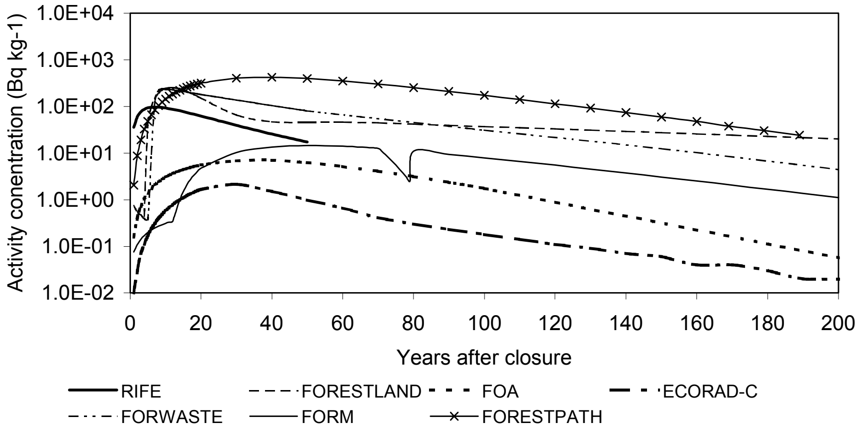


FIG. 43. ^{137}Cs activity concentration in wood without bark (Bq kg^{-1}).

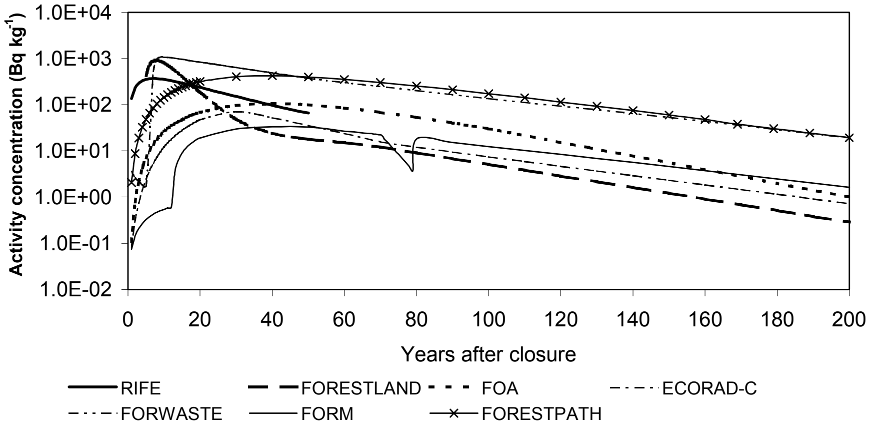


FIG. 44. ^{137}Cs activity concentration in needles (Bq kg^{-1}).

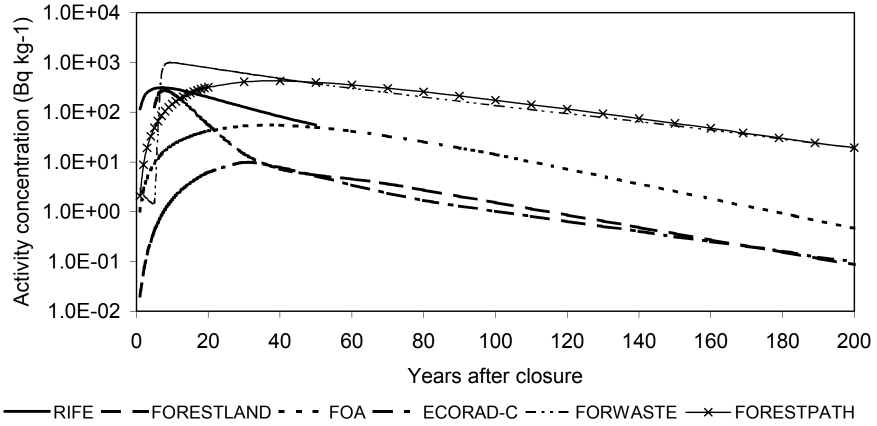


FIG. 45. ^{137}Cs activity concentration in total bark (Bq kg^{-1}).

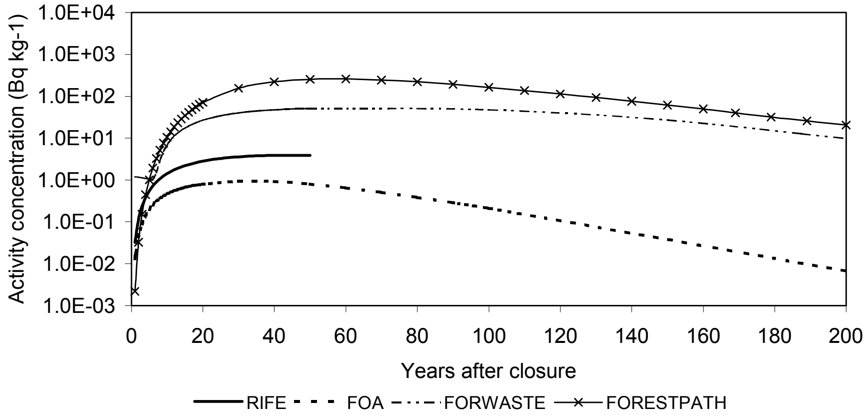


FIG. 46. ^{137}Cs activity concentration soil cover (Bq kg^{-1} dry weight), 0–10 cm from surface.

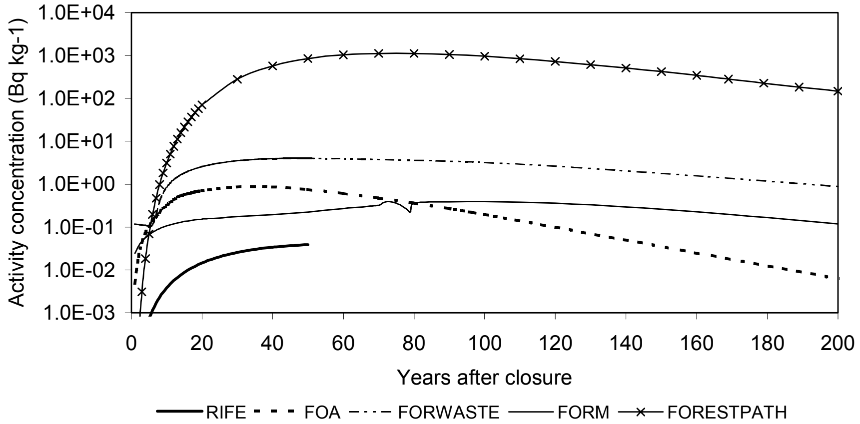


FIG. 47. ^{137}Cs activity concentration in soil cover (Bq kg^{-1} dry weight), 30–40 cm from surface.

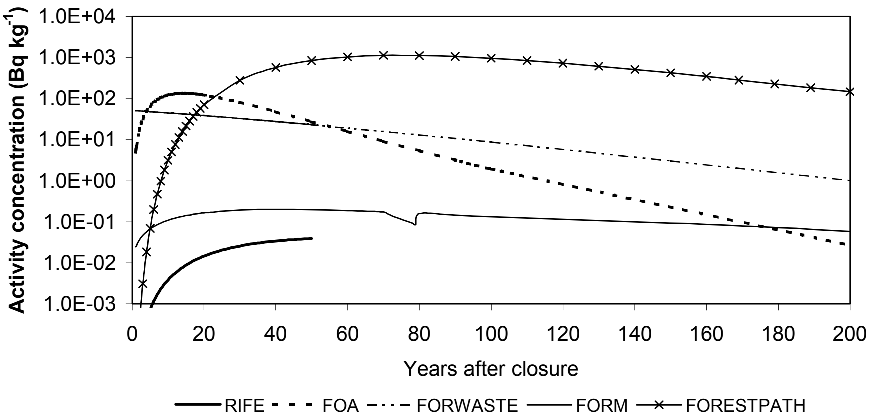


FIG. 48. ^{137}*Cs activity concentration soil cover (Bq kg*$^{-1}$ *dry weight), 90–100 cm from surface.*

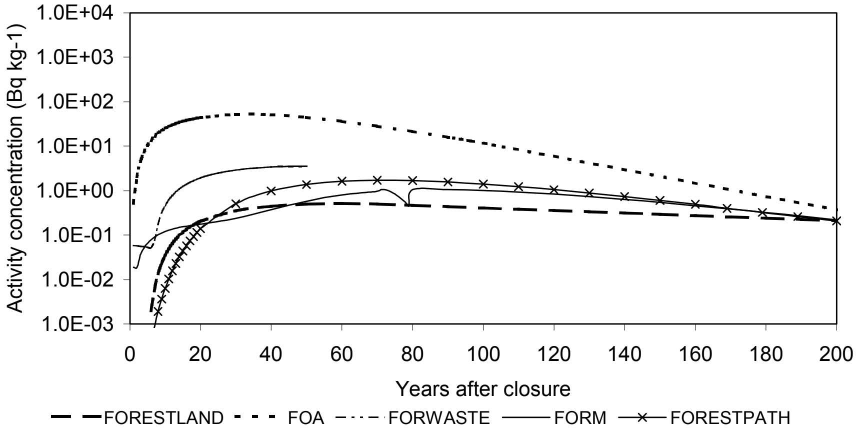


FIG. 49. ^{137}*Cs activity concentration in bilberries (Bq kg*$^{-1}$ *fresh weight).*

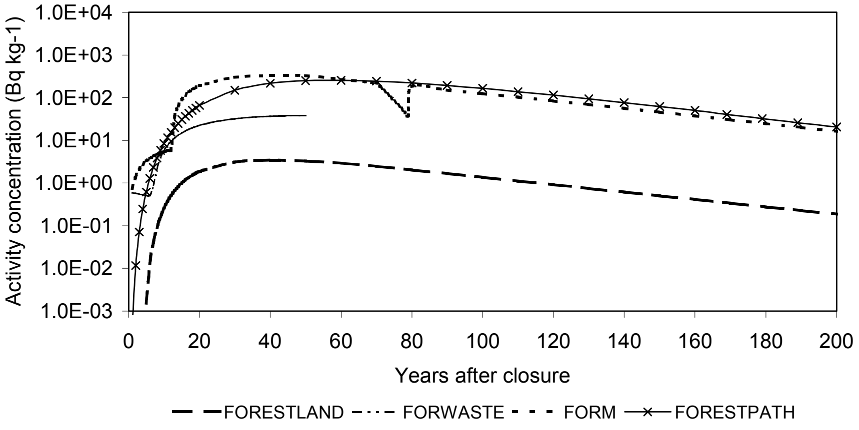


FIG. 50. ^{137}*Cs activity concentration in Xerocomus badius (Bq kg*$^{-1}$ *fresh weight).*

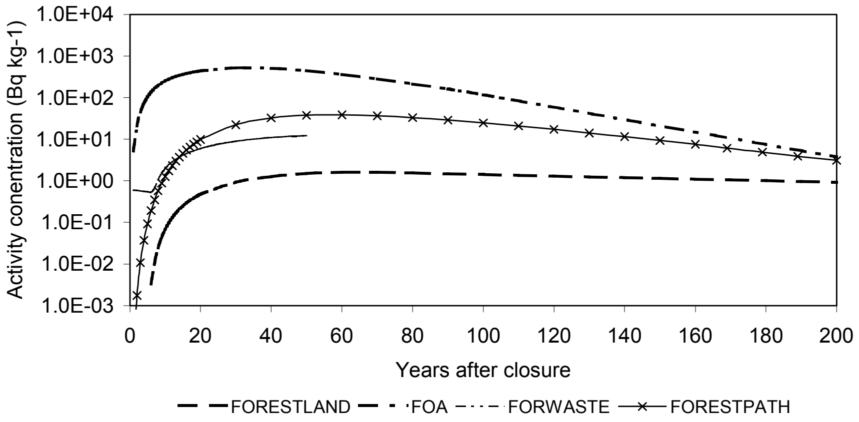


FIG. 51. ^{137}Cs activity concentration in Boletus edulis (Bq kg^{-1} fresh weight).

7.4.1. Trees and associated components

Predictions of activity concentrations of specific components of trees (*Pinus sylvestris*) were provided by all seven respondents, although the FORM model did not return results for the total tree or bark. Figures 42–45 show plotted results for each of the specific tree-related endpoints, namely total tree, wood (without bark), needles and total bark (including cambium).

For each of these endpoints all models except FORESTPATH predicted very similar contamination dynamics over the 50 to 200 year simulation period. Broadly, the consensus of these simulations is that an initially rapid increase in ^{137}Cs activity concentrations in all tree tissues occurs over a period of 10 to 40 years, followed by a steady decline to the end of the 200 year simulation period which can probably be accounted for by radioactive decay. FORESTPATH also predicted a rapid increase in activity concentrations of all tissues up to approximately 50 years following trench capping, but thereafter, despite accounting for radioactive decay, this model predicted a levelling off of the tree activity concentrations, suggesting a balance between tree uptake and radioactive decay.

Examining the time trends predicted by the individual models reveals some interesting nuances within the simulations. The tree uptake dynamics predicted by the FORWASTE model were characterised by an initial decline in ^{137}Cs activity concentration over a period of approximately five years before significant uptake of ^{137}Cs by the tree tissues began. It is unclear why this pattern of uptake emerged but might be due to an assumption that the ‘clean’ soil contained residual ^{137}Cs from atmospheric fallout. The FORM model also produced interesting simulations of activity concentrations in wood and needles in which discrete inflections in the simulated time trends were evident at 10 and 70 years. These times correspond to tree ages at which thinning or harvesting might normally be expected to occur in a managed forest and the FORM model takes this into account. As described above, however, the overall time trend predicted by FORM was similar to that predicted by most of the other models.

In a model–model inter-comparison based on such a novel and hypothetical scenario it is difficult to draw hard conclusions concerning the absolute variability between the predictions of different models. Nevertheless, it is striking that, in general, there was agreement within two orders of magnitude between most models after the commencement of the steady state phase of predictions after 10 to 50 years. This is particularly interesting considering that none of the models had previously been calibrated for this type of scenario and the modellers relied solely on their own judgement when making the necessary modifications to their models to represent, conceptually, the system to be modelled in Scenario 3. Selection of appropriate model parameters also relied on this judgement. A key conceptual problem for all the modellers was how to represent the process of upwards transport of ^{137}Cs through the soil profile and into the trees and vegetation. Solutions to this problem were critical to the simulated vertical distributions of ^{137}Cs which are described in the next section.

7.4.2. Soils

The initial soil profile in Scenario 3 was completely homogeneous in structure, so the scenario description did not include any details of soil horizons. For this reason it was requested that soil distributions of ^{137}Cs were presented for arbitrary 10-cm soil layers. Some models (e.g. FOA and RIFE) only consider a limited number of averaged soil depths, so only selected results produced by the models are considered here. Time course simulations of ^{137}Cs activity concentration at soil depths of 0–10, 30–40 and 90–100 cm are shown in Figures 46–48, respectively. The simulations for the 0–10 cm depth (Figure 46) show very similar time trends but the range of absolute activity concentrations predicted spans up to five orders of magnitude. The envelope of absolute activity concentrations predicted for the mid-section of the soil profile (Figure 47) also spans a similar range of values, but one of the models (FORM) predicts a rather different time trend when compared to the others for which data are available at this depth. Finally, predictions for the deepest part of the originally clean soil profile (90–100 cm, Figure 48) were highly divergent both in the nature of the time trends predicted and the magnitude of soil contamination.

Upwards transport of a relatively highly sorbed radionuclide such as ^{137}Cs can occur as a result of advection–diffusion mechanisms, but also by biological mechanisms such as root uptake and translocation and the bulk turnover of soil by soil fauna (bioturbation). The differences in model simulations of the soil distribution of ^{137}Cs in Scenario 3 may be largely due to different assumptions being made by modellers about the relative importance of these various transport mechanisms. However, though the results presented here suggest that this problem is worthy of further study they do not suggest a clearly preferred methodology for this type of scenario.

7.4.3. Other biological endpoints

The biological endpoints of concern in Scenario 3 were limited to bilberries and two species of ‘mushrooms’. The results for these are shown in Figures 49–51, respectively.

7.4.3.1. Bilberries

Five modellers provided predictions for time courses of contamination in bilberries. The time courses of bilberry contamination predicted by each of these models were similar in both form and magnitude. The contamination of plant species and mushrooms over the long term is principally dependent on soil contamination and, given the conceptual difficulties in modelling soil transport of ^{137}Cs which were identified in the previous section, it is striking that the bilberry predictions are so similar. The main difference between the five models was that, as in the case of tree contamination, FORESTPATH predicted that a plateau of contamination would be reached after some 50 years, whereas the other models predicted a general decline in bilberry activity after 50 to 70 years. The overall discrepancy in absolute predicted activity concentrations was two to three orders of magnitude.

7.4.3.2. Mushrooms'

Predictions were provided by four modellers for *Xerocomus badius* and by four modellers for *Boletus* species, though only three modellers provided predictions for both species. Once again, the general trends of each of the predictions were similar and the overall uncertainty in absolute predicted activity concentrations was approximately three orders of magnitude. This slightly greater uncertainty of the absolute predicted values, compared with those for bilberry, probably reflects the generally greater uncertainty associated with transfer coefficients for 'mushroom' species than for green plants. The similar form of the predicted time trends, again compared with the results for trees and bilberries, no doubt reflects the fact that it is the activity concentrations predicted for the upper soil layer(s) which is the main determinant of contamination of biological endpoints.

7.5. GENERAL SUMMARY

Scenario 3 is dramatically different from both the preceding scenarios and is sufficiently novel, insofar as the modellers' experience is concerned, that the results obtained should be treated with some caution. Nevertheless, the scenario provided some fascinating results that may be very useful in guiding the design and selection of future forest modelling inter-comparison scenarios.

Given the fact that none of the modellers who participated in Scenario 3 has previously had any opportunity to develop or calibrate a model for the uptake and redistribution of ^{137}Cs from a subterranean source, the degree of agreement between predictions for tree components and for biological endpoints has been striking. What is not clear from these results, however, is why there should be such generally good agreement between models. Perhaps the key question to be addressed is the way in which individual modellers represented vertical, upward soil transport. From the selected results shown here for predicted soil distributions there is evidently a degree of dissimilarity in the manner and extent by which soil transport has been approached by each of the models. A key question arising from the results of Scenario 3 is whether physical or biological transport of ^{137}Cs through the soil is likely to dominate a) when the ^{137}Cs is below the surface and b) when deep-rooted plants such as trees are allowed to access such subterranean sources.

8. CONCLUSIONS

8.1. INTRODUCTION

During the period 1998–2000, the BIOMASS Forest WG has addressed a number of issues relevant to the improvement and validation of existing models which are designed to predict the behaviour and fate of radionuclides, principally ^{137}Cs, in forest ecosystems. The Forest WG has provided an active forum within which 11 modelling groups have put forward their models for testing and inter-comparison in three scenarios. In addition, reviews have been undertaken concerning our fundamental understanding of biogeochemical cycling in forest ecosystems, the use of interaction matrices in model design and process identification, and the definition and application of the transfer factor concept in forest ecosystems. The main conclusions from each of these components of the Forest WG work programme are now summarised.

8.2. CONCLUSIONS FROM REVIEWS

- Research into the dynamics of radiocaesium cycling in forests has often been limited to understanding single biogeochemical pathways rather than examining the system *in toto*.
- More data sets addressing radiocaesium cycling within forests on a whole ecosystem basis would be useful to aid model development and validation.
- To achieve the above, there is still a need for standardisation of current approaches to data acquisition in a format which can be used to facilitate the comparison of radiocaesium cycling at different geographical locations and, preferably, on a whole-ecosystem basis.
- An alternative way to improve the development of conceptual models is the application of a systematic method of identifying dominant features, events and processes (FEPs) using an ‘interaction matrix’ approach.
- Carried out correctly, the interaction matrix method should introduce a higher level of objectivity into model design and development.
- When modelling a complex system, such as a forest, the interaction matrix method should enhance the ability of modellers to determine what level of aggregation is optimal and what interactions or pathways should be included.
- Transfer factors, of one type or another, are likely to remain key parameters within dynamic forest models.
- It is impossible to recommend a ‘best option’ for the type of transfer factor which should be used in any particular model since the choice of transfer factors will usually depend on the purpose of the model being constructed.
- Irrespective of the purpose of the model or the type of transfer factor used, however, the precise definition of the transfer factor adopted, and how it is used in a model, should be clearly described by the modeller.
- The definition of transfer factors suitable for application to perennial woody vegetation such as trees remains problematic because the radionuclide burden of wood may have been accumulated over a period of several decades and single TF values may not adequately reflect this.
- A novel method to determine the wood interception potential (WIP) for radiocaesium has been proposed.

8.3. CONCLUSIONS FROM MODEL INTER-COMPARISON EXERCISES

- For the soil and tree compartments examined in Scenario 1 there was generally a high level of consistency between predictions made by the 11 models tested.
- A high level of agreement between predictions for the soil compartments was particularly evident in Scenario 1 and this doubtless reflects the research efforts which have been made over the last 10 years in understanding processes controlling radiocaesium migration in soils.

- Predictions of all the biological endpoints proved more variable, especially predictions for mushroom contamination.
- This raises the question of whether our current (mainly deterministic) modelling approaches are suitable to predict the behaviour of biological entities which will inevitably exhibit a high degree of variability.
- The results of the Scenario 2 inter-comparison generally confirmed those of Scenario 1.
- Particularly accurate and consistent predictions were made for the tree-related compartments and for certain soil compartments at the Rudnya-Povcha site.
- The biological endpoints (roe deer, bilberries and mushrooms), however, proved more difficult to model with certainty.
- A major limitation of the Scenario 2 inter-comparison was that the time-scale for which data were available was limited to the period from 1991 to 1998, which excludes both short-term and genuinely long-term trends.
- There is clearly still a need to keep adding to existing data sets to ensure that the genuinely long term trends of ^{137}Cs distribution in forests are recorded and understood.
- Scenario 3 was dramatically different from the preceding two scenarios and provided a stern test of the ability of modellers to adapt both their conceptual ideas and parameters when considering a subterranean source term of ^{137}Cs.
- The key conceptual question which arose as a result of Scenario 3 was whether vertically upward transport of ^{137}Cs in a soil profile is best considered as a physical process or as a biologically mediated process, especially in the presence of deep-rooting trees.

9. RECOMMENDATIONS AND SUGGESTIONS FOR FUTURE WORK

The following recommendations and suggestions for future work are based on a comprehensive discussion between members of the Forest WG during the final BIOMASS meeting in November 2000. This discussion centred on three main issues, as follows:

- the design and management of model inter-comparison studies;
- improvements in the way radionuclide transfers in forest ecosystems are modelled; and
- suggestions for future work by the BIOMASS Forest WG.

Recommendations and suggestions in each of these categories are presented below.

9.1. RECOMMENDATIONS CONCERNING THE DESIGN AND MANAGEMENT OF MODEL INTER-COMPARISON STUDIES

- Establishment of ground rules for model inter-comparisons. The primary aim of the Forest WG was, from the outset, to conduct model–model and model–data inter-comparisons using available forest radioecology models. While the choice of scenarios was somewhat limited (see next point) it was clear that there were several different ways in which the inter-comparisons could be designed and implemented. The manner in which the three inter-comparisons reported in Sections 5, 6 and 7 of this report were conducted is not presumed to be optimal, but in the absence of any generally accepted guidelines for such exercises it was difficult to judge how best to manage each scenario. Questions arose within the Forest WG as to whether all modellers should make predictions for all endpoints, should model–data inter-comparisons be 'double blind' (which effectively excludes a modeller who has a suitable data set on which to base an inter-comparison scenario since he/she might already have used those data to calibrate their model), how many modelling iterations should be allowed and should modellers be allowed to make second-round predictions after seeing the data set they first tried to model unseen? These and several other questions lead to the recommendation that a generally accepted set of ground rules should be established (or at least agreed upon by any group of modellers wishing to compare and test their models) to simplify the management of model inter-comparison studies and to assist in quality assurance of these exercises.

- Choice of 'challenging' scenarios. One general criticism of the three inter-comparisons conducted by the Forest WG was that they were too similar to scenarios with which most of the modellers were familiar. In other words, the chosen scenarios were not challenging enough. Scenario 3 (Section 7) was the most challenging, involving a sub-surface source term which none of the modellers had previously addressed. However, even this scenario prescribed a forest type which had been the focus of most of the modellers' previous efforts. It is recommended, perhaps as part of establishing a set of ground rules as described above, that scenarios are designed which test the available models to the reasonable limits of their performance.
- Provision of single 'calibration' datum points in blind model–data inter-comparisons. One of the problems in conducting a truly blind model–data test is that the modellers receive no feedback on the performance of their model until they see the full data set after submission of their simulation. Provision to the modellers of a single datum point from the blind data set as part of the inter-comparison scenario has several advantages. The main advantage for the modeller is that he/she can determine how well his/her model is performing with respect to that datum point, thereby avoiding order of magnitude discrepancies between data and model predictions. This benefits both modeller and the inter-comparison in general since it ensures that discrepancies between the performance of individual models are not due to 'wildcard' errors in the calibration of individual models for the specific scenario being considered. The further benefit for the inter-comparison is that, even if all models 'predict' the single calibration datum perfectly, the simulated kinetics of each model before and after that datum point can be compared to provide valuable information on the discrepancies between simulations. Selection of a suitable datum point from a blind data set which may contain tens or even hundreds of data is a difficult question, but it is recommended that such a datum is provided when conducting model–data inter-comparisons.
- The need to give estimates of uncertainty in model simulations. Despite the request for modellers to provide estimates of 95% uncertainty bounds around predictions made in each of the three model inter-comparisons conducted by the Forest WG, only one model (FORESTPATH) consistently provided such estimates. Both Type A (stochastic) and Type B (system/process) uncertainties contribute to the overall uncertainty of a model simulation. To some extent, the potential for Type B uncertainties in individual models can be gauged by comparing the representation of forest compartments and transfer processes in that model against an idealised system representation, such as that shown in the matrix in Figure 2, although this comparison is qualitative. It is, however, becoming increasingly straightforward to represent stochastic uncertainties of parameters in

models and it is recommended that co-ordinators of model inter-comparison exercises make it mandatory for participants to provide uncertainty estimates, even if this dictates that scenarios are kept simple to avoid overloading participating modellers.

- Model versus modeller uncertainty. An interesting question is that of model versus modeller uncertainty. This was not a real issue in the Forest WG since each modeller was using a model which he/she had developed themselves. The question arises, however, if two modellers are each using the same model independently of each other. This question has been addressed in detail by Linkov and Burmistrov [2001].
- Model–data comparisons are most useful. It was the generally agreed conclusion of the Forest WG that scenario 2, the model–data inter-comparison, proved to be the most interesting and the most useful test of model performance. This is in accordance with the conclusions of Hoffman and Hofer [1988], who concluded that the applicability of model–model inter-comparisons was questionable because this method "offers no measure of accuracy without independent test data". It is recommended that, where possible (i.e. when genuinely independent and unseen data sets are available), model–data inter-comparisons should be conducted in preference to model–model inter-comparisons.
- Are forest models and modellers fit for purpose? Since the inter-comparison exercises addressed by the Forest WG did not deal with specific applications (e.g. dose assessment, evaluation of countermeasure effectiveness) the question of 'fitness for purpose' of participating models is difficult to answer. However, recommendations for future studies in Section 9.3 include the suggestion that specific case studies be addressed. It is recommended that such studies be pursued to determine whether the current generation of forest radioecology models is fit for purpose.

9.2. RECOMMENDATIONS CONCERNING MODELLING OF RADIONUCLIDE CYCLING IN FOREST ECOSYSTEMS

- Time dependency in forest models should include tree growth. One of the fundamental characteristics of trees is that their biomass changes with time. This change is generally referred to as 'growth', which implies an increase in biomass with time although, in the later stages of a tree's life cycle, its biomass can decline significantly. The complex natural cycle of biomass change in forests is closely linked to elemental cycling, especially the cycling of carbon which constitutes the major proportion of biomass of trees. Models of ecosystem development proposed by authors such as Aagren and Bosatta [1996] incorporate growth as an integral and fundamental component of the models. However, in many current radioecological models growth is not explicitly (or even implicitly) represented – no mention is made of tree growth in Figure 2, for instance! It is recommended that in any future BIOMASS exercises involving forests that tree biomass be considered as a process of fundamental radioecological significance.
- High versus low contamination scenarios. It was evident after the Chernobyl accident that differences in the initial deposited activity of a radionuclide can vary by several orders of magnitude, especially if the near-field and far-field are compared. The point was raised during the Forest WG discussions that almost all the models participating in the inter-comparison exercises represent transfer processes as linear phenomena, hence differences in the total activity burden within a forest make no difference to the model

simulations. Over the potential order-of-magnitude range of activity concentrations, however, it is possible that this assumption is invalid. Furthermore, the existence of large concentration ranges of naturally occurring analogues such as ^{133}Cs and K for ^{137}Cs, ^{88}Sr and Ca for ^{90}Sr, can lead to non-linear behaviour, especially during biological uptake and sorption in soils. One of the Forest WG models (FORSUN) represents competition between the Cs and K ions during soil sorption as a non-linear process. However, it is recommended that other modellers begin to represent key mechanisms in their models which might be inherently non-linear

- Process-oriented models are preferable for long-term predictions. Leading on from the last point, it was concluded that process-orientated (i.e. mechanistic) models, which would include key non-linear processes such as diffusion, are preferable for long-term predictions since site-specific calibration for the far future is impossible. This approach would also facilitate the inclusion of other pollutants in forest models since the use of empirical transfer functions and coefficients would be avoided as much as possible. This approach is recommended, but the practical difficulties involved are acknowledged and the reader is referred to the discussion of transfer factors in Section 3 of this report.
- Integration of forest radioecology models with models for other ecosystems. With the advent of improved software for modelling and the ability to integrate models with Geographical Information Systems, it is recommended that forest radioecology models be designed in future as part of general landscape models. The consideration of ecosystems such as forests and adjacent pasture or aquatic systems as separate ecological entities is artificial, though greatly simplifies the modeller’s task. However, for the future development of forest radioecology models it would be preferable, and particularly useful for specific assessment problems such as radioactive waste disposal, to focus at a landscape level.
- Is it possible to construct a generic model for any ecosystem based on fundamental ecological properties? This final and challenging question was raised by the Forest WG. It is well established that variations in ecosystem type between climatically distinct locations, for instance, are due to variations in primary production, biological decomposition rates etc. The ability to model radionuclide behaviour based on such fundamental properties of an ecosystem would enable the model to be scaled to warm/cold and wet/dry climates. This characteristic would help to address the potential effects of climate change on radionuclide behaviour in the environment, especially over the long time scales which need to be addressed in the context of waste disposal.

9.3. SUGGESTIONS FOR FUTURE WORK BY THE BIOMASS FOREST WORKING GROUP

The following list of suggestions is proposed on the basis of detailed discussion within the Forest WG, although only brief discussions and justifications for each suggestion are provided here:

- Long-term and retrospective simulations of radionuclide behaviour in forests. As already discussed above, certain assessment problems, such as radioactive waste disposal, demand that long-term simulations of radionuclides in natural ecosystems such as forests are made. The validity of model simulations over periods of several decades could be determined using retrospective case studies for which calibration data could be obtained for the present day. One good candidate for such a study would be the forested area affected by the Kyshtym accident in 1957.

- The problem of forest edges. Forest edges are important since it is known that enhanced wet and dry deposition occurs here and they are often exploited for food by man and animals in a significantly different way than the 'deep' forest. Forest edges are common and extensive landscape features and consideration of forest edges might necessitate the landscape approach described in Section 9.2.
- The geosphere/forest interface. An important interface of significance to the problem of migration of radionuclides from underground repositories is the geosphere/forest interface. Scenario 3 (Section 7) made a start in addressing the role of forest vegetation in enhancing the migration of radiocaesium from the sub-surface to the surface soil horizons. However, further consideration of this type of scenario might assist the work of BIOMASS Theme 1.
- The effect of forest fires on radionuclide distribution. Forest fires exert an extremely important influence in the long-term development of forests world-wide, although specific information on the effects of such fires on re-distribution of radionuclides is almost non-existent.
- Application of existing forest models to case studies such as countermeasure/ remediation effectiveness and establishment of reference levels for forest products such as timber/timber products.
- Integration of Forest WG activities with those of BIOMASS Themes 1 and 2. The relevance of forest modelling to the problem of radioactive waste disposal has been mentioned in Sections 9.1 and 9.2. Dose reconstruction activities, especially for accidents such as Kyshtym and Chernobyl, could also benefit from the inclusion of the expertise of the Forest WG members.

REFERENCES

ÅGREN, G.I., BOSATTA, E., Theoretical ecosystem ecology: understanding element cycles, Cambridge University Press (1996).

AVILA, R., MOBERG, L., A Systematic Approach to the Migration of Cs-137 in forest ecosystems using interaction matrices, J. Environ. Radioactivity **45** (1999).

AVILA, R., MOBERG, L., HUBARD, L., Modelling of radionuclide migration in forest ecosystems, SSI Report **98:07** ISSN 0282-4434 (1998).

RAVILA, A., Radioesium in the forest and forest industry: Studies on the flow, occurrence and technological enhancement of radionuclides in the forest with emphasis on the behaviour of radiocesium, Dept. of Radiation Physics, Lund University Hospital (1998).

BARCI-FUNEL, G., DALMASSO, J., BARCI, V.L., ARDISSON, G., Study of the transfer of radionuclides in trees at a forest site, Sci. Total Environ. 173/174 (1995) 369–373.

BELLI, M., (Ed.), Long-term dynamics of radionuclides in semi-natural environments: derivation of parameters amd modelling, ANPA, Rome (2000).

BONNETT, P.J.P., ANDERSON, M.A., Radiocesium dynamics in a coniferous forest canopy: A Mid-Wales study, Sci. Totoal Environ. **136** (1993) 259–277.

BRECHINAC, F., MOBERG, L., SUOMELA, M., Long-term environmental behaviour of radionuclides. CEC-IPSN Association Final Report, IPSN, Fontenay-aux-Roses, France (2000).

BYRNE, A.R., Radioactivity in fungi in Slovenia, Yugoslavia, following the Chernobyl accident, J. Environ. Radioactivity **6** (1988) 177–183.

COLE, D.W., VAN MIGROET, H., Chronosequences: A technique to assess ecosystem dynamic, In: Research strategies on long-term site productivity, Proc. IAE/BE A3 Workshop, Seattle WA, August 1988, FRI Bulletin **152** (1989) 5–24.

COLE, D.W., RAPP, M., Elemental cycling in Forest Ecosystems, In: Reichle D.E (Ed) Dynamic properties of forest ecosystems (International Biological Programme; 23), Cambridge University Press, Cambridge (1981) 341–409.

DELVAUX, B., MAES, E., THIRY, Y., KRUYTZ, N., Fate of radiocaesium in forests and forest soils: a review, In: Wenzel et al. (eds), Proc. of “Biogeochemistry of Trace Elements”, 5th ICOBTE ‘99, Vienna, Austria, July 11–15 1999, **1** (1999) 66–67.

DELVAUX, B., KRUYTS, N., CREMERS, A., Rhizospheric mobilization of radiocesium in soils, Environ. Sci. Technol. **34** (2000) 1489–1493.

DESMET, G.M., VAN LOON, L.R., HOWARD, B.J., Chemical speciation and bioavailability of elements in the environment and their relevance to radioecology, Sci. Total Environ. **100** (1991) 105–124.

ERTEL, J., ZIEGLER, H., Cs-134/137 contamination and root uptake of different trees before and after the Chernobyl accident, Radiat. Environ. Biophys. **30** (1991) 147–157.

FAWARIS, B.H., JOHANSON, K.J., Radiocesium in soil and plants in forest in central Sweden, Sci. Total Environ. **157** (1994) 133–138.

FRISSEL, M.J., An update of the recommended soil-to-plant transfer factors of Sr-90, Cs-137 and transuranics, VIIIth Report of the Working Group Soil-to-Plant Transfer Factors, IUR (1992).

GUILLITTE, O., FRAITURE, A., LAMBINON, J., Soil–fungi radiocesium transfers in forest ecosystems, In: Transfer of Radionuclides in Natural and Semi-Natural Environments (eds. G. Desmet, P. Nassimbeni, and M. Belli), Elsevier Applied Science, London, New York (1990) 468–476.

HAAS, G., SCHUPFNER, R., MULLER, A., Radionuclide uptake and long-term behaviour of Cs-137, Cs-134 and K-40 in tree rings of spruce, Journal of Radioanalytical and Nuclear Chemistry Articles **194** 2 (1995) 277–282.

HOFFMAN, F.A., HOFER, E., An overview of the IAEA Safety Series on procedures for evaluating the reliability of predictions made by environmental transfer models, In "Reliability of Radioactive Transfer Models", ed. G. Desmet, Elsevier Applied Science, London and New York (1988) 1–14.

HORRILL, A.D., KENNEDY, V.H., HARWOOD, T.R., The concentrations of Chernobyl derived radionuclides in species characteristic of natural and semi-natural ecosystems, In: Transfer of Radionuclides in Natural and Semi-Natural Environments (eds. G. Desmet, P. Nassimbeni, and M.Belli), Elsevier Applied Science, London, New York (1990) 27–39.

INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Radionuclide Release into the Environment: Assessment of Doses to Man (ICRP Publication 29), Annals of the ICRP 2 (1979).

Derivation of quantitative acceptance criteria for disposal of radioactive waste to near surface facilities: development and implementation of an approach, IAEA draft report version 3.0, March 1999.

INTERNATIONAL ATOMIC ENERGY AGENCY, Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments, Technical Reports Series No. 364, IAEA, Vienna (1994).

IPATYEV V.A., BRAGINSKY V.F., BULAVIK I.M., et al., "Forest. Man. Chernobyl", Gomel, (1999) (in Russian) 454 p.

LINKOV, I., SCHELL, W.R. (Ed), Contaminated Forests: Recent Developments in Risk Identification and Future Perspectives. NATO Science Series, Kluwer Academic Publishers (1999).

LINKOV, I., BURMISTROV, D., Model or Modeller Uncertainty: Lessons Learned from the IAEA Model Intercomparisons (2001).

LINKOV, I., Radionuclide Transport in Forest Ecosystems: Modelling Approaches and Safety Evaluation, PhD Thesis, University of Pittsburgh (1995) 171 p.

MAMIKHIN, S.V., TIKHOMIROV, F.A., SHCHEGLOV, A.I., Dynamics of 137-Cs in the forests of the 30-km zone around the Chernobyl nuclear power plant, Sci. Total Environ. **193** (1997) 169–177.

MCGEE, E.J., JOHANSON, K.J., KEATINGE, M.J., SYNNOTT, H.J., COLGAN, P.A., An evaluation of ratio systems in radioecological studies, Health Phys. **70** (1996) 215–221.

MYTTENAERE, C., SCHELL, W.R., THIRY, Y., SOMBRE, L., RONNEAU, C., VAN DER STEGEN DE SCHRIECK, J., Modelling of the Cs-137 cycling in forests: recent developments and research needed, Sci. Total Environ. **136** (1993) 77–91.

NIMIS, P.L., Radiocesium in plants of forest ecosystems, Stud. Geobot. **15** (1996) 3–49.

NYLÈN, T., Uptake, turnover and transport of radiocaesium in boreal forest ecosystems. Thesis, Swedish University of Agricultural Sciences, Uppsala, Sweden (1996).

RANGER, J., BONNEAU, M., Effets prévisibles de l'intensification de la production et des récoltes sur la fertilité des sols de forêts. Le cycle biologique en forêt. R.F.F., vol. XXXVIII, **2** (1984) 93–111.

RIESEN, T.K., AVILA, R., MOBERG, L., HUBBARD, L., Review of forest models developed after the Chernobyl NPP accident, In: I. Linkov and W.R. Schell (Eds), Contaminated forest, Kluwer Academic Publishers, The Netherlands (1999) 151–160.

RÜHM, W., KAMMERER, L., HIERSCHE, L., WIRTH, E., The ^{137}Cs/^{134}Cs ratio in fungi as an indicator of the major mycelium location in forest soil, J. Environ. Radioactivity **35** (1997) 129–148.

RÜHM, W., STEINER, M., KAMMERER, L., HIERSCHE, L., WIRTH, E., Estimating future radiocaesium contamination of fungi on the basis of behaviour patterns derived from past instances of contamination, J. Environ. Radioactivity **39** (1998) 129–147.

RÜHM, W., YOSHIDA, S., MURAMATSU, Y., STEINER, M., WIRTH, E., Distribution patterns for stable ^{133}Cs and their implications with respect to the long-term fate of radioactive ^{134}Cs and ^{137}Cs in a semi-natural ecosystem, J. Environ. Radioactivity **45** (1999) 253–270.

SCHELL, W.R., LINKOV, I., MYTTENAERE, C., MOREL, B., A dynamic model for evaluating radionuclide distribution in forests from nuclear accidents, Health Phys. **70** (1996) 318–335.

SCHELL, W.R., THIRY, Y., SEYMOUR, E.M., Modeling of Radionuclides and Trace Elements in Forest Ecosystems, In I. Linkov and W.R. Schell, (eds.): Proc. of "Contaminated Forests: Recent Developments in Risk Identification and Future Perspectives". NATO Advanced Research Workshop, Kiev, Ukraine, June 24–28 1998, Kluwer Academic Publishers, Netherlands, NATO ASI Series, 2. Environmental Security **34** (1999) 255–261.

THIRY, Y., KRUYTS, N., DELVAUX, B., Respective horizon contributions to Cesium-137 in Soil-to-Plant transfer: A Pot Experiment Approach, J Environ. Qual. **29** (2000).

THIRY, Y., Étude du Cycle du Radiocésium en Ecosystème Forestier: Distribution et Facteurs de Mobilité, PhD Thesis, Faculté des Sciences Agronomiques, Université Catholique de Louvain, Louvain-la-Neuve, Belgium (1997).

THIRY, Y., MYTTENAERE, C., Behaviour of radiocaesium in forest multilayered soils, J. Environ. Radioactivity **18** (1993) 247–257.

TIKHOMIROV, F.A., SHCHEGLOV, A.I., SIDOROV, V.P., Forests and forestry: radiation protection measures with special reference to the Chernobyl accident zone, Sci. Total Environ. **137** (1993) 289–305.

TSUKADA, H., SHIBATA, H., SUGIYAMA, H., Transfer of radiocaesium and stable caesium from substrata to mushrooms in a pine forest in Rokkasho-mura, Aomori, Japan, J. Environ. Radioactivity **39** (1998) 149–160.

ULRICH, B., Influence de la fertilisation sur le cycle des éléments nutritifs dans les écosystèmes forestiers, C.R. Congrès Paris 1973, IUFRO/FAO (1973) 23–34.

WILLIAMS, L.R., LEGGETT, R.W., A measure of model reliability, Health Physics **46** 1 (1984) 85–95.

WIRTH, E., KOEHLER, H., BURKHARDT, J., The suitability of transfer coefficients used for stochastic calculations in radioecology, Health Phys. **49** (1985) 1165–1172.

YOSHIDA, S., MURAMATSU, Y., Concentrations of alkali and alkaline earth elements in mushrooms and plants collected in a Japanese pine forest, and their relationship with ^{137}Cs, J. Environ. Radioactivity **41** (1998) 183–205.

Further reading:

GILLETT, A.G., CROUT, N.M.J., A review of ^{137}Cs transfer to fungi and consequences for modelling environmental transfer, J. Environ. Radioactivity **48** (2000) 95–121.

ANNEX I
MODEL DESCRIPTIONS

I–1. RIFE1

G. Shaw
Centre for Analytical Research in the Environment,
Imperial College, Centre for Environmental Technology,
Silwood Park, Sunningdale, Ascot, Berkshire, United Kingdom

The RIFE1 model has been used in this submission. The structure is shown in Figure I-1.

I–1.1. GENERAL

The five white compartments represent dynamic state variables in the forest system. Fluxes of radionuclides introduced into these compartments are represented as series of couple d first order differential equations which are solved numerically and which account for the mass balance of radionuclides within the system.

The shaded compartment represents biological entities which can be considered to be in quasi-equilibrium with the system over the medium- to long-term. Accordingly, aggregated transfer coefficients (T_{agg}) are used to calculate radionuclide activity concentrations in this compartment. This compartment has been used to represent mushrooms (fungi), understorey (berries) and game animals in the calculations carried out for this scenario.

I–1.2. TREES

Pine trees only have been considered in these calculations. Tree growth has been accounted for in the simulations, using the growth data provided in the scenario description.

I–1.3. SOIL

RIFE1 allows simulations of radionuclides in litter, organic soil and mineral soil. Characteristics of litter were given separately in the scenario description. Organic soil in the RIFE1 simulations combines AoF and AoH horizons. For the mineral soil, only the AoA1 and A1 horizons have been considered — this is an arbitrary choice.

I–1.4. FUNGI, UNDERSTOREY AND ANIMALS

As described above, T_{agg} values have been used to calculate activity concentrations in fungi, understorey and animal compartments. These have been obtained from the IAEA booklet of recommended values (Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments, Technical Reports Series No. 364). Ranges of values are presented in this handbook and ‘best estimates’ have been calculated from these ranges for the purposes of these simulations.

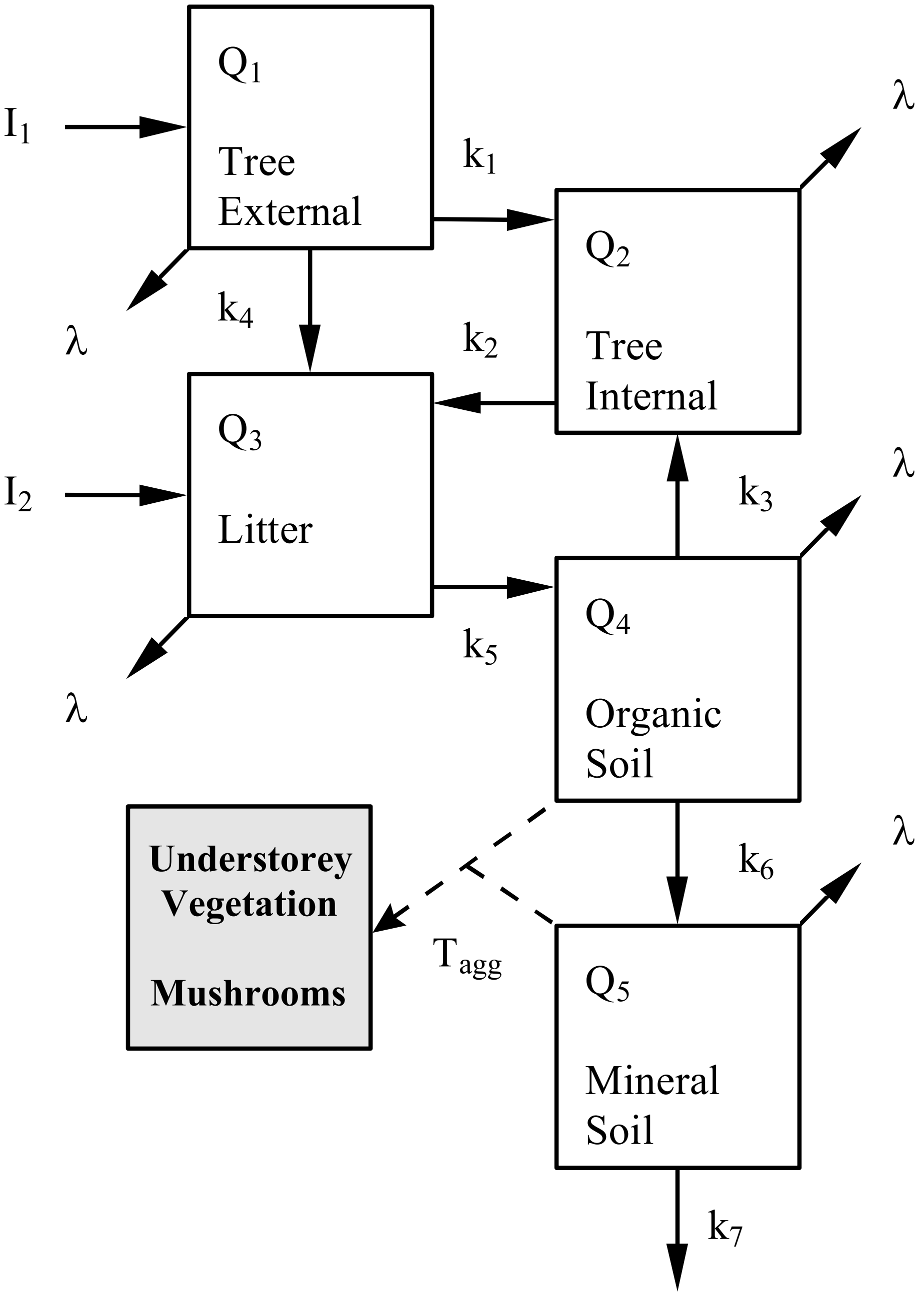


FIG. I-1. Structure of the RIFE1 model.

I–2. FORESTLAND

R. Avila
Swedish Radiation Protection Institute (SSI),
Stockholm, Sweden

S. Fesenko, S. Spiridonov
Department of Radioecology,
Russian Institute of Agricultural Radiology,
Obninsk, Russian Federation

FORESTLAND is a dynamic model for the prediction of temporal and spatial patterns of the consequences of radioactive contamination of forests ecosystems. The model is focused on migration pathways leading to internal and external radiation doses to the population. FORESTLAND can be applied to both the acute and long-term phases of the contamination created from an aerial radioactive deposition. The present version of the model consists of five individual models:

- FORBIO: A model of the biomass dynamics of trees and the understorey vegetation;
- FORGAME: A dynamic model of the long-term migration of radionuclides in forest food chains, including wild animals;
- FORACUTE: A dynamic model of the migration of radionuclides in forest ecosystems during the acute phase of the contamination;
- FORTREE: A model of the long-term migration of radionuclides in forest trees;
- FOREXT: A dosimetric model for calculation of gamma dose rate in the forest.
- FORDOSE: A model for calculation of the internal and external doses to the population (presently under development).

The structure of FORESTLAND, where outputs of one model are inputs to other models (Figure I-2), is specified below.

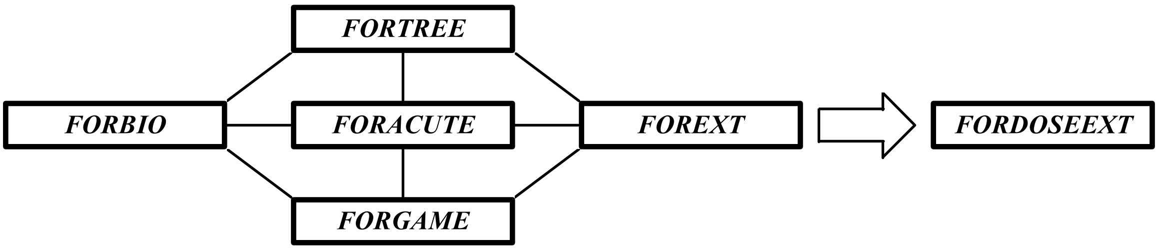


FIG. I-2. Interconnection of the individual models within FORESTLAND.

Interconnections between the individual models:

- The model FORBIO provides the migration models (FORTREE, FORGAME and FORACUTE) with input parameters for calculation of several transfer rates. The values of biomass density, in units of kg m^{-2}, estimated with FORBIO are used to obtain the concentration of radionuclides in different forest components, in units of Bq kg^{-1}, from the values of the radionuclide content in these components, in units of Bq m^{-2}, calculated with the migration models.

- The initial distribution of radionuclides in different forest components needed in FORTREE and FORGAME (initial conditions) are calculated with FORACUTE. Alternatively the user can define directly the initial conditions.
- FORACUTE and FORGAME provide the values of activity levels in different forest components needed for calculation of the dose rates with FOREXT. A similar connection between FORTREE and FOREXT (dashed arrow in Figure I-2) is being implemented. FORBIO provides FOREXT with some parameters needed for calculation of the attenuation and scattering of gamma rays.
- Part of the input needed by FORDOSE is generated by the migration models and FOREXT.

A classification of forest ecosystems into four different categories has been adopted in FORESTLAND. Each category corresponds to a different type of tree (coniferous or deciduous) and landscape (automorphic or hydromorphic). A set of model parameters, consisting of a best estimate value and an interval of variation, is estimated for each forest category. A scale of values is defined for each model parameter, which reduces the uncertainties of the parameter values selected for each specific application of the model.

In FORBIO a simple approach for describing seasonal and long-term biomass dynamics of trees and understorey vegetation have been applied. For the understorey vegetation and mushrooms, biomass growth is simulated with a logistic model, while an exponential decrease is assumed during senescence. Differentiation is made between summer and autumn mushrooms and between fruits of berries and the whole plant (animal feeds).

The biomass growth of an individual tree is also described with a logistic model while an exponential equation is used for calculation of tree mortality. A linear differential equation, obtained by combining the equations for growth and mortality, is used for simulating the long-term changes of tree biomass density (kg m^{-2}).

For tree leaves (needles) distinction is made between seasonal and long-term biomass dynamics. The yearly values of leaves (needles) biomass depend of the age of the tree. It is assumed that the contribution of leaves (needles) to the total tree biomass decreases from 10–15% for a 15 years old tree to 1–2% for a 100 years old tree. The seasonal variation of the leaf biomass is described with a logistic model during the periods of growth and senescence.

FORACUTE is a dynamic model of the migration of radionuclides during the acute phase of the contamination, lasting a few years after an aerial deposition. The model describes the primary interception of aerial deposited radionuclides by the above ground vegetation and their subsequent redistribution by transfer processes like weathering, secondary interception, translocation in the tree and the understorey vegetation, and root uptake from the upper soil–litter layer. The model also permits evaluating the dynamics of the radionuclides levels in forest products consumed by man, including forest game.

To describe the primary and secondary retention of the radionuclides by the above-ground phytomass, the latter is viewed as a set of four successive filters: the tree leaves (needles), the tree bark, the understorey vegetation and the upper soil–litter layer. The interception by the understorey vegetation is calculated with an exponential function of the biomass density (Chamberlain’s equation). A method similar to the one commonly used for evaluating the passage of light though tree crowns is used to simulate the initial retention of radionuclides by trees. It is assumed that the initial retention by trees is proportional to the "projective cover

(PC)" of the tree crows. The PC can be calculated from the crown closure (relative area of crowns) and the crown tracery coefficient, which depends on the tree species.

The model FORESTGAME is a dynamic model to predict seasonal and long-term changes of ^{137}Cs activity concentrations in forest food changes (Figure I-3). The mathematical formulation of FORESTGAME corresponds with the so-called linear compartment models. A set of 20 coupled differential equations describes the net accumulation of the radionuclide in the compartments over time. Since the model is focused on forest food chains, the migration in tree is described in a simpler way than in FORTREE. The transfer rates corresponding to the processes of root uptake and translocation in trees are, for instance, described with ordinary rate constants. The soil on the contrary is modeled in more detail (18 compartments) with the purpose of describing the influence of roots and mycelia location on root uptake by the understorey vegetation and mushrooms.

Root uptake by the understorey vegetation is described as a function of root distribution in soil, available fraction of ^{137}Cs in soil, the soil-to-plant concentration ratio and the biomass growth rates. The CRs are related only to the available fraction of the radionuclide in soil and have, therefore, the same values for all soil–litter layers.

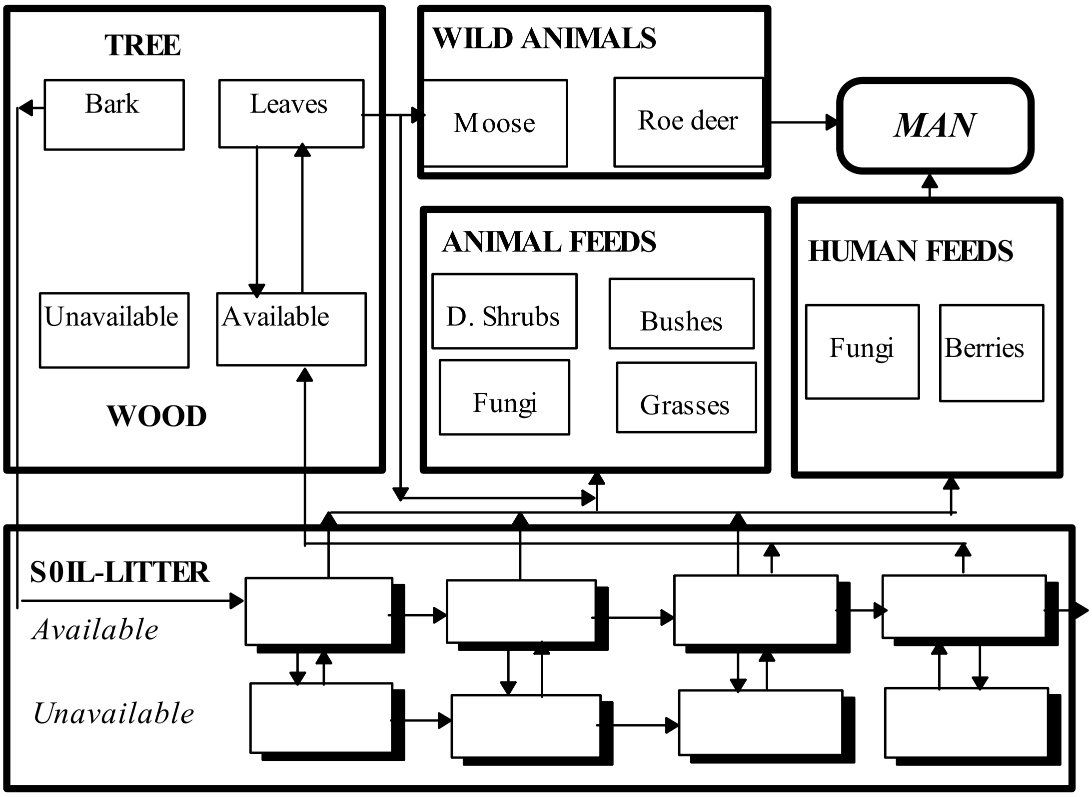


FIG. I-3. Conceptual scheme of the FORESTGAME model.

The intake rate of ^{137}Cs by wild animals (roe deer and moose) is described with a function of the total feed intake, the share of different feeds in the daily animal diet and the activity concentrations in different feeds. It is assumed that the radionuclides incorporated by the animal via ingestion are instantly distributed in the animal body and that the elimination rate from muscles (edible meat) is proportional to the activity levels in this part of the animal body.

FORTREE is a dynamic compartment model of the migration of radionuclides in the tree. The main purpose of the model is interpretation and prognosis of the long-term kinetics of ^{137}Cs activity concentrations in wood. Another endpoint of the model is the seasonal change of activity concentrations in leaves (needles). The basic diagram of the FORTREE conceptual model for deciduous and coniferous trees is presented is presented in Figure I-4.

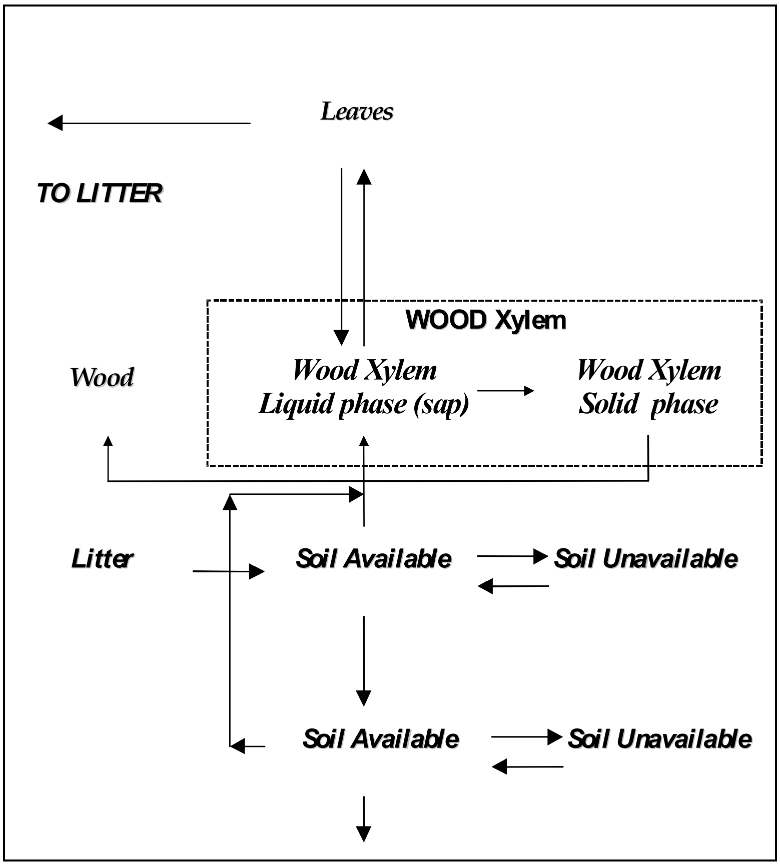
Five compartments are used to describe the change of the quantity of available ^{137}Cs in soil. The first soil layer (0–10 cm) contains the most active parts of the root and is responsible for root uptake during the first several decades after the radioactive contamination. Each soil layer is divided into an available and an unavailable fraction with the purpose of considering sorption/desorption processes leading to fixation and remobilization of the radionuclides.

The tree is described with the following compartments: two compartments for the living part of the wood (liquid and solid phases of the wood xylem), one compartment for the dead (structural) wood, one compartment for leaves (deciduous trees) and four compartments for needles of different age class (coniferous trees).

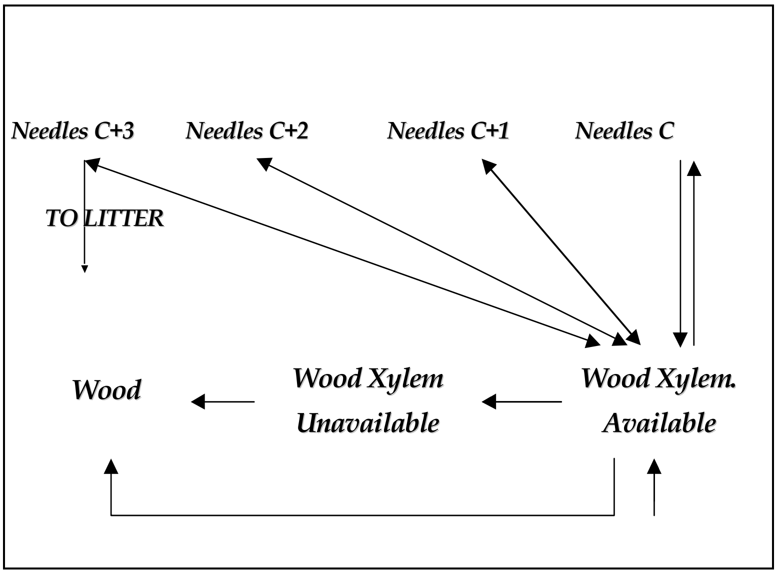
Root uptake is a function of the water flux (transpiration flow) through the xylem during the vegetative period. The main part of this flux is due to evaporation from leaves, which is proportional to the leaves (needles) biomass. The root uptake rates are, therefore, expressed as a function of the leaves (needles) biomass, which depends on tree biomass and age. A conceptual scheme of the FORTREE model is given in Figure I-4.

FOREXT is a dosimetric model for calculation of the dose rates in forests contaminated with gamma emitters. Although FOREXT can be used to calculate the exposure rates at any height and for any gamma emitter, in FORESTLAND the model is only adjusted for estimation of the dose rate from ^{137}Cs at 1 m above the soil surface. The vertical column of the forest is divided into 7 successive layers with different average densities. The soil is described with four layers, one for each soil horizon (L, Of, Oh and A). The fifth layer goes from the soil surface to the average height of the understorey vegetation. The frontier between the sixth and seventh layer is set at the average height of the bottom of the tree canopy.

Each layer is considered as a plate source of finite thickness. The activity of each source is calculated from the values of ^{137}Cs activity concentrations in different components of the layer provided either by FORGAME or FORACUTE. The variation of these values with time leads to time variation of the estimated dose rates. The attenuation and scattering of the photons in different layers is taken into account in the dose rate calculations.



Deciduous forest



Coniferous forest

FIG. I-4. Conceptual scheme of the FORTREE model.

I–3. FOA

R. Bergman
National Defence Research Establishment,
Department of NBC Defence,
Umeå, Sweden

I–3.1. THEORETICAL PRESENTATION OF CAESIUM BEHAVIOUR IN A COMPLEX FOREST ECOSYSTEM

Our model (Bergman et al. 1993), focused on redistribution processes in a long-term perspective, belongs to the explanatory category. In our model the major regulators of energy flow, as well as of caesium turnover, are related to primary production and its constraints on the growth capacity. Certain fundamental physiological processes governing the metabolism of living matter in the biotope are also considered. The principal structure of the model is shown in Figure I-5.

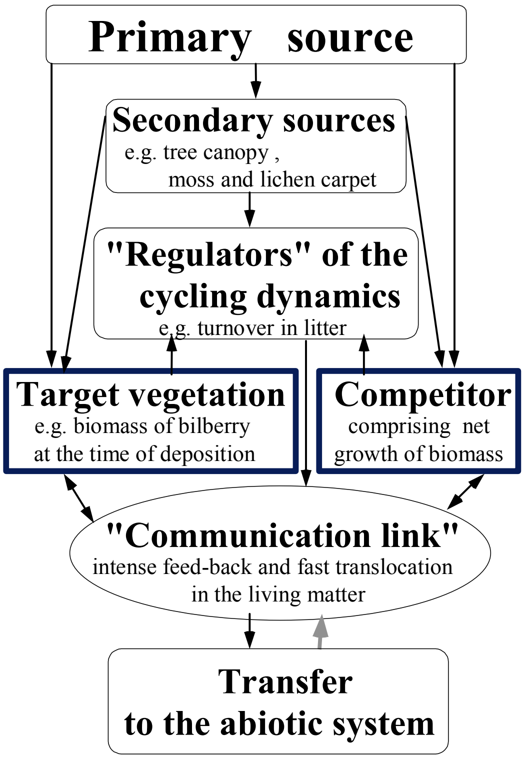


FIG. I-5. The principal model structure for interactions and turnover of ^{137}Cs in boreal forest ecosystems.

The model includes qualitatively effects of primary production and growth on turnover of caesium. The dependence on these factors is concluded from the following facts: primary production and its distribution over growth and litterfall constitute major regulators with regard to the dynamics of the redistribution processes of organic matter in the forest. The same conditions should be true for redistribution effects on potassium due to its essential role in the living cell. Potassium and caesium are to a high degree exchangeable in active transport over cell membranes in living tissue. Evidently both elements may serve in the same vital processes. Accordingly, as primary production is of importance for the behaviour of potassium in the forest ecosystem, it should be so for caesium too.

The qualitative system structure of processes, interactions and compartments is thus mainly based on physiological characteristics concerning transport of caesium over cell membranes and intracellular distribution, and the apparently conservative conditions prevailing for caesium in boreal ecosystems (Bergman 1994). Also, quantitative estimates have been made from the latter conditions — e.g. the fact that very little of the radioactive caesium deposited over the forest area is lost from the system by run-off, about 90% of the total deposition of ^{137}Cs occurs in the upper organic horizon in podzol areas, and that the availability in the ecosystem, as can be seen from the ^{137}Cs concentration in moose meat, was not significantly different in 1985 (i.e. prior to the Chernobyl accident) compared to the period 1986–1990 (Bergman et al. 1991).

The theoretical analysis is based on compartment theory and first order kinetics for the turnover of caesium in a boreal forest. The calculated time dependent change of the ^{137}Cs content in perennial vegetation has been compared to that actually observed at different local study sites with the focus particularly on bilberry.

The primary purpose of applying this model has been to elucidate qualitatively how predictions based primarily on growth and physiological behaviour of caesium corroborate with the main features of the time-dependent change of ^{137}Cs activity according to measurements on perennial vegetation.

I–3.1.1. Redistribution and transfer processes

Estimated transfer factors (Bergman et al. 1993) are based on the actual results for the time-dependent redistribution of ^{137}Cs from secondary sources in a Scots pine canopy by throughfall and needlefall (Nylén and Grip 1989, Nylén 1996), in addition to the release to the environment of ^{137}Cs deposited over the moss and lichen carpet. After the Chernobyl accident loss from the system by runoff is less than that due to physical decay — from 1987 and onwards — and is therefore disregarded in the model. The model also includes: a “competitor” compartment (i.e. indicating the increase in biomass competing for the available caesium) to simulate influence on the redistribution processes of primary production and growth; *target vegetation* (i.e. the biomass of the perennial vegetation under study at the time of deposition); litterfall from this compartment; decomposition in a litter compartment; and exchange of caesium between the vegetation compartments and soil. See Bergman et al. 1993 for detailed list of transfer factors and model parameters.

Effects of growth

Essential factors for the site specific growth dynamics are:

- Maximum attainable total biomass;
- Dynamics of age dependent net productivity;
- Successional stage and age of the forest stand.

Both in short and long-term perspectives these factors are expected to regulate the redistribution of caesium within and between the biotic components of the system. At sites with a poor nutrient state (e.g. on peat soil) the net biomass increase is very limited, implying relatively small "dilution" effects on the concentration of ^{137}Cs in the vegetation by redistribution of some fraction of it to the new biomass. Similarly in an old forest, where the biomass already has approached rather closely to the maximum capacity of the site productivity, further net increase in biomass is limited — i.e. only relatively small changes in the concentration of ^{137}Cs may be expected from growth and subsequent dilution. At sites with good soil conditions possible to support a high biomass, on the other hand, forests at young stages generally exhibit a fast net increase in biomass, which is expected to influence the concentration of e.g. ^{137}Cs considerably in the vegetation. Growth functions adapted to simulate the dynamics of net growth representative for many sites in the boreal vegetation zones are illustrated in Figure I-6. Growth of the *competitor* compartment (cf. Figure I-5) is governed by this time-dependence and scaled to the appropriate level of maximum biomass associated with the soil conditions at the particular sites under study.

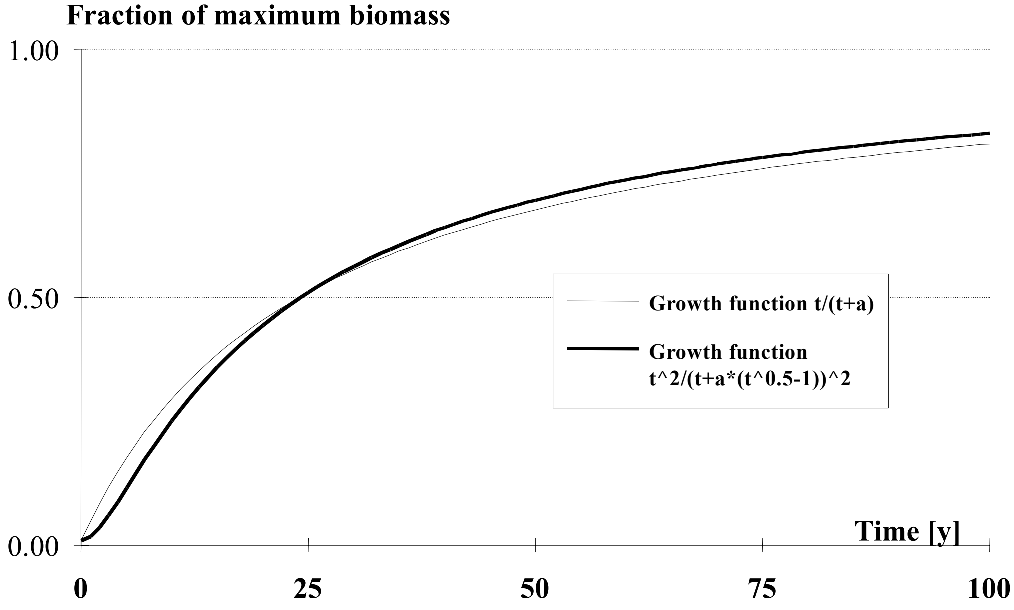


FIG. I-6. Fraction of maximum biomass.

I–3.1.2. Model predictions

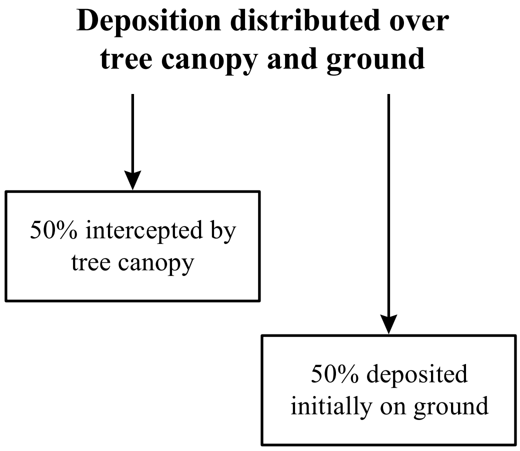


FIG. I-7. (cf. Bergman et al. 1993).

The behaviour of deposited ^{137}Cs has been simulated using age of the forest at fallout and nutrient state of the growing site as parameters. Interception in the tree canopy has been chosen to be similar to that which resulted after the Chernobyl accident from wet deposition at the study site (Bergman et al. 1988, Nylén and Ericsson 1989, Nylén 1996) and also in coniferous forests at several sites elsewhere in the boreal zones (Bergman1994).

References

BERGMAN, R., DANELL, K., ERICSSON, A., GRIP, H., JOHANSSON, L., NELIN, P., NYLÈN T., Uptake, turnover and transport of radioactive nuclides in a boreal forest ecosystem, FOA rapport E 40040 (1988) (in Swedish).

BERGMAN, R., NYLÈN, T., PALO, T., LIDSTRÖM, K., The behaviour of radioactive caesium in a boreal forest ecosystem, In: The Chernobyl fallout in Sweden — Results from a research programme on environmental radiology, Moberg, L. (Ed), Swedish radiation protection institute, 425–456 (1991).

BERGMAN, R., NYLÈN, T., NELIN, P., PALO, T., Caesium-137 in a boreal forest ecosystem: Aspects on the long-term behaviour, Det sjette Nordiske Radio¢kologi Seminar, 1992, Tùrshavn Fêroyar, FOA report C 40284-4.3 (1993).

BERGMAN, R., The distribution of radioactive caesium in boreal forest ecosystems, In: Nordic Radioecology: The transfer of Radionuclides through Nordic Ecosystems to man, Dahlgaard, H. (Ed), Elsevier Sci. Publ. (1994).

NYLÈN, T., ERICSSON, A., Uptake and retention of Cs-137 in Scots Pine, Proc. XVth Congress of IRPA, Progress in radiation protection, Feldt, W. (Ed) (1989).

NYLÈN, T., GRIP, H., The Origin and Dynamics of Cs-137 Discharge From a Coniferous Forest Catchment, accepted in J. Hydrology (1995).

NYLÈN, T., Uptake, turnover and transport of radiocaesium in boreal forest ecosystems, Thesis, Swedish University of Agricultural Sciences, Uppsala, Sweden (1996).

I–4. FORESTLIFE

A. Dvornik, T. Zhuchenko
Laboratory of Radioecology,
Forest Institute of the National Academy of Sciences of Belarus,
Gomel, Belarus

I–4.1. SUMMARY

The phenomenologic model FORESTLIFE was created to predict radionuclide concentrations in compartments of the forest ecosystem based on the idea that a forest ecosystem is a living organism with its usual laws of the nutrition, growth and circulation of chemical elements and energy. The parameters of the model were determined using the database obtained as the result of full-scale regular observations of the radionuclide distribution in compartments of the forest ecosystem on the network of forest plots. The mathematical dependence between necessary parameters were obtained from experimental data. The model permits to predict the radionuclide contamination levels for all components of forest stands of different types. The validation of model was carried out by the comparison of its results with the experimental data of the other independent measurements and the results of other models.

I–4.2. STRUCTURE OF THE FORESTLIFE MODEL

The FORESTLIFE model uses the following model approaches:

- the ^{137}Cs vertical migration along the soil profile is described by two-component quasi-diffusion and convective-diffusion models. Dynamics of the root area contamination is calculated through the use of these model parameters,
- the model considers a single radionuclide fallout,
- the model is used to estimate the level of contamination of a tree only by the root uptake of radionuclides,
- the ratios of plant-available ^{137}Cs compounds at different depth of the root zone are considered to be time independent but dependent on forest soil types,
- the rate of ^{137}Cs uptake in trees decreases with its age. In the model, the age dependence of the ^{137}Cs transfer factor into wood is considered to be time dependent.

The structure of the FORESTLIFE model is illustrated schematically in Figure I-8.

References

IPATYEV, V., BULAVIK, I., BAGINSKY, V., GONCHARENKO, G., DVORNIK, A., Forest and Chernobyl, Forest ecosystems after the Chernobyl nuclear power plant accident, 1986–1994, Journal of Environmental Radioactivity, **42**, Elsevier Science Limited (1999) 9–38.

DVORNIK, A., ZHUCHENKO, T., Model FORESTLIFE and prediction of radioactive contamination of forests in Belarus, In: Contaminated Forests: Resent Developments in Risk Identification and Future Perspectives, I. Lynkov, W.R. Schell (Eds), Kluwer Academic Publishers, Netherlands (1999) 185–194.

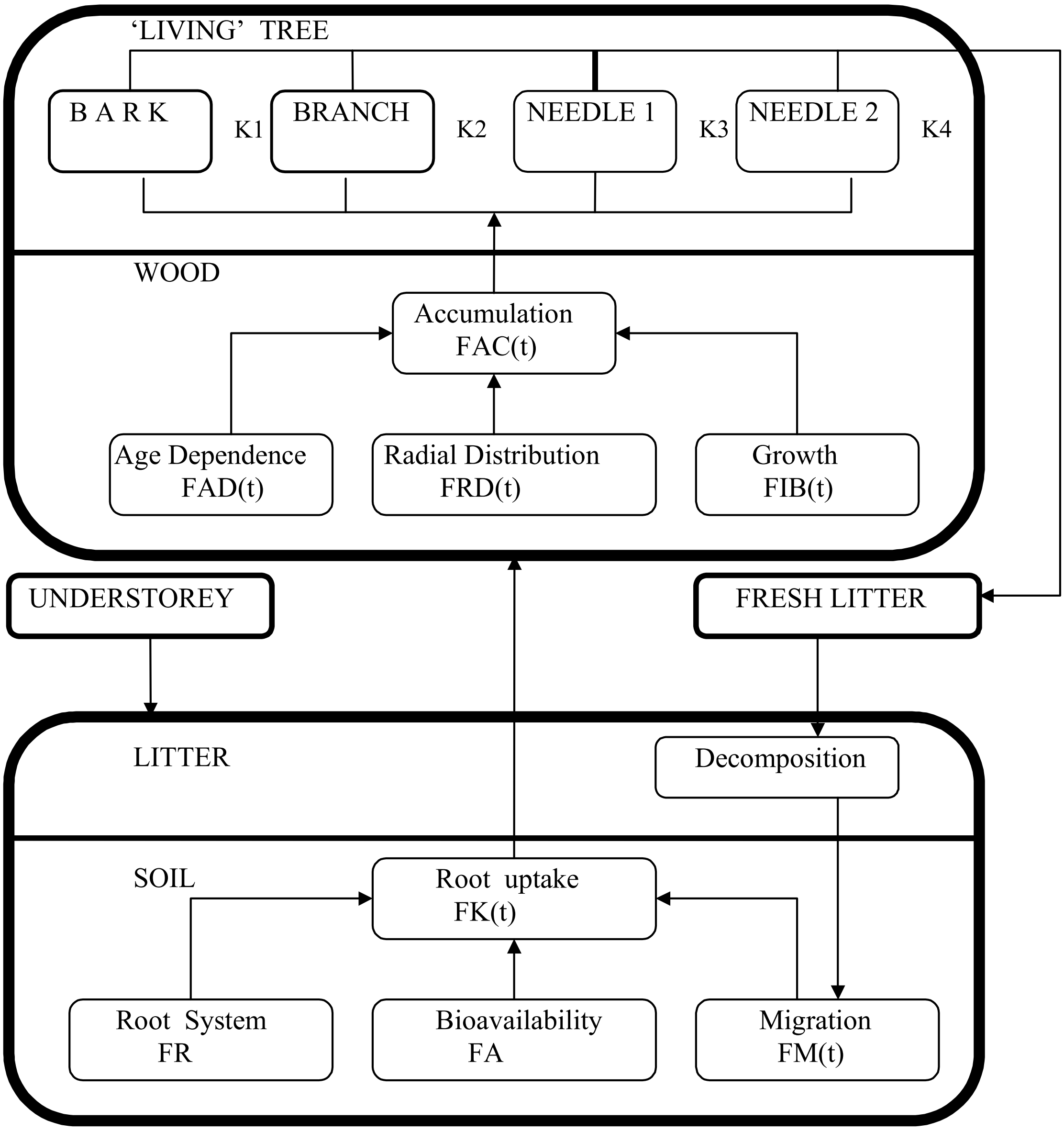


FIG. I-8. The structure of the FORESTLIFE model.

I–5. FORESTPATH

I. Linkov
Menzie-Cura and Associates, Inc.,
Chelmsford, MA, United States of America

Linkov , 1995 and Schell et al., 1996a developed the generic model for radionuclide transport in forests, FORESTPATH, which calculates time series of inventories for a specific radionuclide distributed within the following eight compartments: Understorey, Tree, Organic Layer, Labile Soil, Fixed Soil and Deep Soil. To incorporate details of radionuclide migration in the Organic Layer and fungi, the model was developed further (Figure I-9). The Organic Layer was represented by three horizons: Ol (litter), Of and Oh (Schell et al., 1996c, Linkov et al., 1999).

In the initial studies (Schell et al., 1996a, Schell et al., 1996b) uncertain model parameters were estimated for the generic model application from the literature, and satisfactory model predictions provided a general view of radionuclide fate and transport. For site-specific applications, the available literature data were limited to the ecosystems close to the site under consideration; site-specific parameters were thus estimated. Nevertheless, this deterministic approach has a limited site-specific application because it does not provide uncertainty estimates for the radionuclide concentrations in the compartments. Therefore, it cannot be used to estimate the confidence intervals for radiation doses required in risk assessment.

In recent studies (Linkov et al., 1997, Linkov at al., 1999) the model uncertainty is treated probabilistically. Results of a literature review show that values for model parameters are very uncertain and can be presented only by broad probability distributions. A triangular shape for the distributions is assumed, characterized by three parameters: minimal and maximal values, and mode.

References

LINKOV, I., Radionuclide Transport in Forest Ecosystems: Modeling Approaches and Safety Evaluation, PhD Dissertation, University of Pittsburgh (1995).

LINKOV, I., MOREL, B., SCHELL, W.R., Remedial policies in radiologically-contaminated forests: environmental consequences and risk assessment, Risk Analysis 17 67 (1997).

LINKOV, I, BURMISTROV, D., KANDLIKAR, M., SCHELL, W.R., Reducing Uncertainty in the Radionuclide Transport Modeling for the Chernobyl Forests Using Bayesian Updating in: Linkov, I., W.R. Schell, Eds, “Contaminated Forests: Recent Developments in Risk Identification and Future Perspectives”, Kluwer, Amsterdam (1999).

SCHELL, W.R., LINKOV, I., MYTTENAERE, C., MOREL, B., A dynamic model for evaluating radionuclide distribution in forests from nuclear accidents, Health Physics 70 318 (1996a).

SCHELL, W.R., LINKOV, I., RIMKEVICH, et al., Model-directed sampling in Chernobyl forests: general methodology and 1994 sampling program, Science of the Total Environment 180 229 (1996b).

SCHELL, W.R.; LINKOV, I., BELINKAIA, E., MOREL, B., “Application of a dynamic model for evaluating radionuclide concentration in fungi”, In: Proc. of 1996 Int. Congress on Radiation Protection, Vienna (1996c) 2–752.

TABLE I-1. FORESTPATH MODEL COEFFICIENTS USED AS ^{137}Cs INPUT FOR CONIFEROUS FOREST IN THE IAEA BIOMASS SCENARIO

Parameter	Notation	Distribution	Parameters	
			mode	a–b*
absorption half-time (years)	t_{ab}	triangular	0.6	0.2–1
tree biomass (t/ha)	B_t	uniform	140	120–160
desorption half-time (years)	t_{ds}	triangular	1.1	0.5–10
leaching half-time (years)	t_{lc}	triangular	800	350–3000
organic layer removal half-time (years)	or	8	1–100	3
Ol	_ol		0.6	
Of	_of		1.3	
Oh	_oh		1.7	
radiation half-time (years)	$t_{1/2}$	constant	30.14	na
tree uptake half-time (years)	t_{tu}	uniform	10	1–100
tree removal half-time				
short (<1 weeks)	_tr	triangular	3.6 d	1.4–5 d
intermediate (<1 yr)			80 d	21–175 d
tree removal half-time long (>1 yr)	t_{tr}		3	1–10
understorey removal half-time				
short (< 1 weeks)	_ur		12 d	9–19 d
intermediate (<1 yr)			32 d	8–64 d
long (>1 yr)	t_{ur}	triangular	0.2	0.1–2.8
understorey uptake half-time (years)	t_{uu}	uniform	8	1–100

*a is the minimum value and b is the maximum value for triangular and uniform distributions.

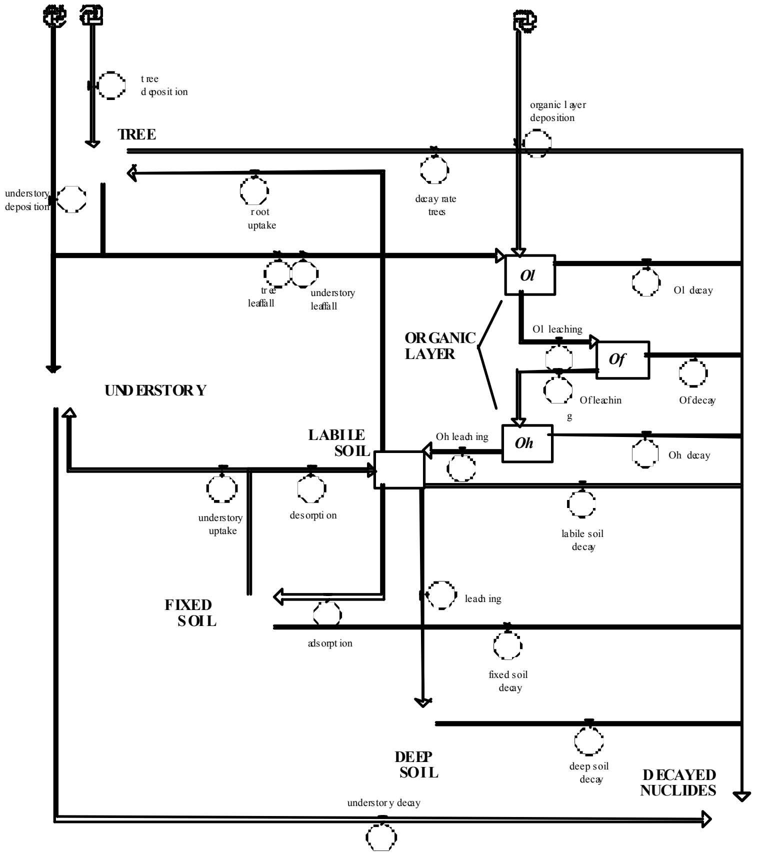


FIG. I-9. Diagrammatic representation of the FORESTPATH model.

I–6. ECORAD-C

S. Mamikhin
Moscow State University,
Moscow, Russian Federation

The behaviour of the radionuclide is assumed to obey the same regularities as the behaviour of its stable chemical analogue — potassium. Radionuclide dynamics is considered in parallel with the dynamics of phytomass. Radionuclides contained in the vegetation are pooled into two basic compartments: external and internal contamination, with separate analysis of each one. The model was verified using the data obtained during 1986–1994 in the 30 km zone of the accident on Chernobyl NPP.

I–6.1. DRIVING VARIABLE

Atmosphere — the arrival of radioactive matter from the atmosphere.

I–6.2. STATE VARIABLES

The following state variables are included in the model:

Organic matter content (absolutely dry weight) – Xi; content of stable potassium (^{39}K) – Ki; content of ^{137}Cs (Bq on the kilogram of absolutely dry weight): Zi – internal contamination, Yi – external contamination, Ei – total contamination. Soil contamination – Scd. Total contamination of plant and soil cover – Cd. Index i corresponds to the structural parts of plants: 1– distributive pool, 2 – needles, 3 – branches, 4w – trunk wood, 4b – trunk bark, r – large roots, rsm – small roots.

I–6.3. AUXILIARY VARIABLES

Auxiliary variables bell and sig are included to account for the influence of some factors in equations of transfer functions. Variable bell(a,b,c,d) is a Pirson's curve of I type. It describes an asymmetric dependency of a process from the value of factor a and is bell shaped. It is equal to 0 under a<b or a>d and takes a maximum value 1 under a = c. Parameter e causes a widening of the bell shape. Second variable, given by the formula:

sig(a,b,c) = (a–b)/((c–b)/2–b)+(a–b)), increases from 0 under a<b to 1 under a>c.

Since seasonal (within a year) dynamics is not considered in the given variant of the model, the inventory of organic matter in the components is computed using the following equation: Xi(t+1) = Xi(t) + Pi – Oi, where Xi(t+1) – the biomass of given year, Xi(t) – the biomass of previous year, Pi – the gross increment for a year, Oi – the annual litterfall. Then, potassium inventory in the structural components of the plant cover (Ki) and gross increment (Kpi) are computed: Ki = Xi * Ci; KPi= Pi * Ci, where Ci is the specific content of potassium (g/g of absolutely dry matter) in the component i.

I–6.4. TRANSFER FUNCTIONS

I–6.4.1. External contamination

Interception of ^{137}Cs by needles, branches and trunk bark: f0i = a(i–1) * (1 – sig(psize, 0, 1000)) * Xi * finp, where a(i–1) is the retaining ability factor, psize is the size of particles (in micrometers) , Xi is the inventory of organic matter in the component, and finp is the contamination density (kBq km^{-2}). Function sig reflects the inverse dependence of fraction retaining ability on the size of fallout particles.

Litterfall: fyis = bi * (Yi – fyi1), where bi is the litterfall factor (the parameter reflecting the rate of self-decontamination of the fraction from the external contamination).

Radioactive decay: fyid = dc * yi, where dc – part of ^{137}Cs, decaying per a year.

The contribution of structural parts to the distributive pool: fyi1 = ci * Yi, where ci is the proportion of external ^{137}Cs contained in the fraction i entering the pool.

I–6.4.2. Internal contamination

Deductions of ^{137}Cs from fractions into the distributive pool: fzi1 = hi * Zi, where hi is the part of internal ^{137}Cs contained in the fraction i entering the pool. The distribution of ^{137}Cs from the pool into the fractions is assumed to be directly proportional to the K content in the fraction: fy1zi = Y1 * Ki / Ksum, where Ksum is the total K content in vegetation. ^{137}Cs uptake by plants from soil:

fsp = a6 * a7* bell(ny, 1, 5, 100, 5) * Scd,

where a6 is the factor of the ecosystem moistening (hydromorphism), a7 is the factor of maximum biological availability of ^{137}Cs for some type of soil, bell is the function depending on the dynamics of ^{137}Cs biological availability on the time passed from the moment of fallout. Distribution of ^{137}Cs entered from soil into plants by fractions: in this case it is assumed that the distribution takes place proportionally to the content of potassium in the gross increment of some fraction: fsi = fsp * KPi / KPsum, where KPsum is the total content of K in the gross increment of vegetation. Removal of the accumulated ^{137}Cs from the plants to soil with litterfall: this value is assumed to be directly proportional to the litterfall mass: fis = Zi * Oi / Xi. The export of a proportion of ^{137}Cs into the distributive pool before the litterfall is taken into account for the needle compartment.

TABLE I-2. PARAMETER VALUES

		Units
a1	0.141	$m^2\,kg^{-1}\,y^{-1}$
a2	0.089	$m^2\,kg^{-1}\,y^{-1}$
a3	0.056	$m^2\,kg^{-1}\,y^{-1}$
a4	0.9	y^{-1}
a5	0.22	y^{-1}
a6	1	y^{-1}
a7	0.0017	y^{-1}

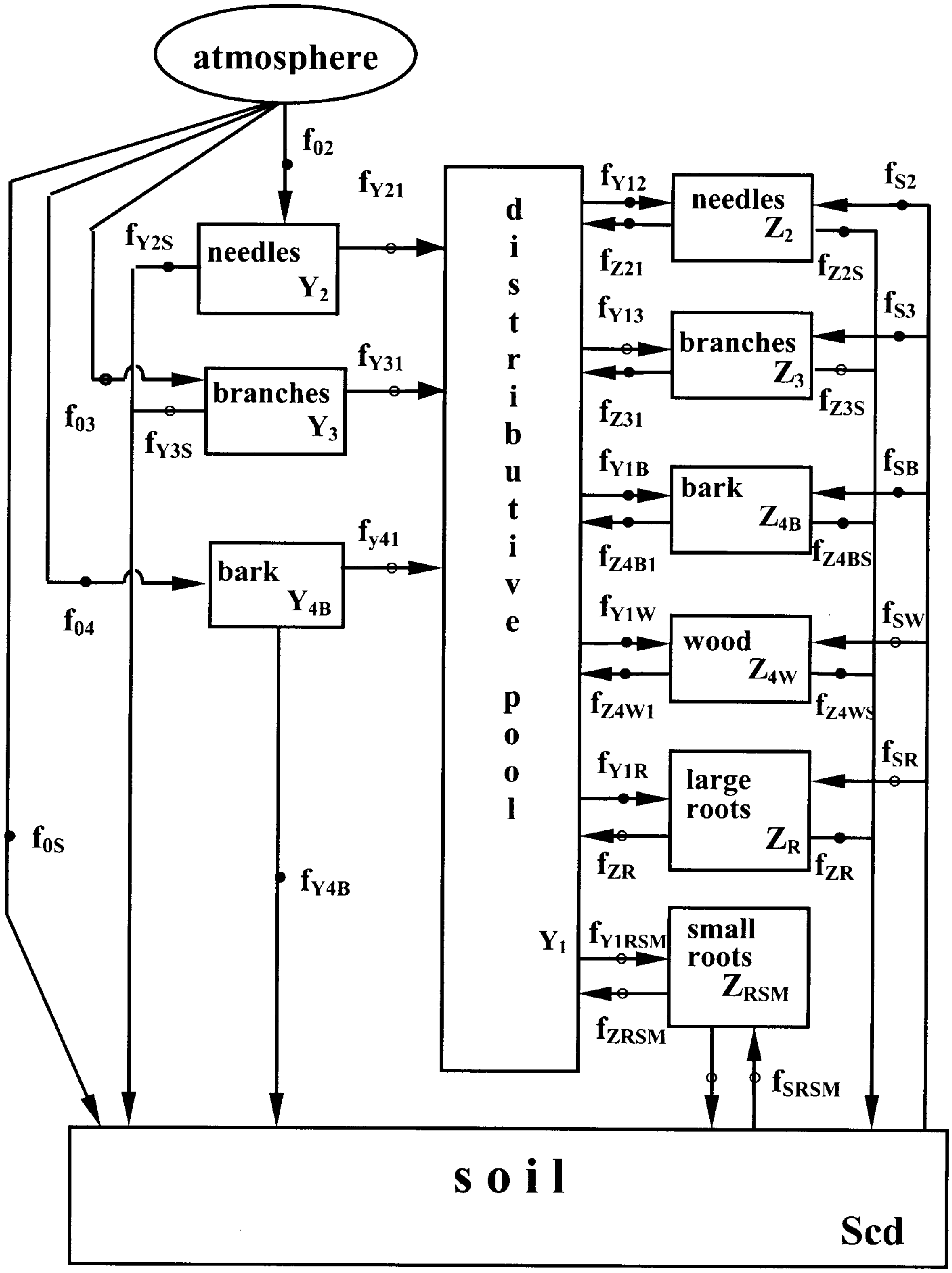


FIG. I-10. Flow diagram of ECORAD-C.

I–7. FINNFOOD

A. Rantavaara
Research and Environmental Surveillance, Ecology and Foodchains,
Radiation and Nuclear Safety Authority (STUK),
Helsinki, Finland

The model for biological endpoints was based on observed constancy of uptake of radiocaesium by forest food products during the post-Chernobyl period in north European conditions rather similar to those defined for the scenario. The ^{137}Cs concentrations have been calculated using the formula:

$$^{137}Cs\ concentration = (Deposition\ in\ ground\ layer\ corrected\ for\ radioactive\ decay) \times transfer\ factor\ (TF)$$

The concentrations in all food products are given for the date August 1. The activity content of trees was not considered.

Transfer factors, TF ($m^2\,kg^{-1}$ fw)	
Moose	0.01
Roe deer	0.05
Boletus edulis	0.0045
Cantharellus cibarius	0.012
Leccinium scabrum	0.021
Russula sp.	0.06
Suillus luteus	0.014
Fragaria vesca	0.004
Rubus idaeus	0.0014
Vaccinium myrtillus	0.0048

TF values are derived from the data of STUK, except for roe deer, which was taken from the IAEA handbook of parameter values (Technical Reports Series No. 364).

I–8. RODOS

A. Rantavaara, J. Wendt
Research and Environmental Surveillance,
Radiation and Nuclear Safety Authority (STUK),
Helsinki, Finland

P. Calmon
IPSN/DPRE/SERLAB/LMODE,
CE Cadarache,
Saint-Paul-lez-Durance, France

Food and Dose Module for Forests (FDMF) integrated after revisions in RODOS version PV4.0

I–8.1. SOFTWARE

RODOS (Real time On-line DecisiOn Support) is a software system for giving decision support for off-site emergency management in Europe. RODOS consists of subsystems for estimation of present and future distributions of activity concentrations, for quantifying benefits and drawbacks of various combinations of protective actions and countermeasures, and for evaluation of countermeasure strategies. The subsystem for estimation of activity concentrations consists of foodchain and dose modules for terrestrial and aquatic ingestion pathways. Currently, a module for modeling forest foodchains and doses, FDMF, is being developed in collaboration of IPSN and STUK. After completion, the module was integrated into the RODOSversion PV4.0.

The Foodchain and Dose Module for Forest ecosystems is composed of four submodules, for deposition, forest dynamics, external exposure and ingestion doses received through mushrooms, berries and game. Deposition calculation can be carried out for fifteen nuclides out of 64 at a time. The ingestion dose calculation is performed for nine radionuclides (I-131, Cs-134, Cs-137, Sr-89, Sr-90, Pu-238, Pu-239, Pu-240, Pu-241).

The main input parameters of this model are the time integrated concentration of radionuclides in air ($Bq.s.m^{-3}$), for dry deposition, and the total wet deposition ($Bq.m^{-2}$) and rainfall. Three forest types with different characteristics (heights, biomass and tree species) may be defined by the user. Calculations are performed for arbitrary geographical regions in Europe. The main output quantities are the concentration of radionuclides in forest gifts (mushrooms, berries and game) ($Bq\ kg^{-1}$ fresh weight), the external exposure to forest workers and to the public, and the ingestion dose for the average population, mushroom and berry collectors as well as hunters.

I–8.2. MODELS

I–8.2.1. Deposition modelling

Dry deposition to all forest compartments is calculated from time integrated concentration in air using deposition velocities and leaf area indices for vegetation compartments (crown, trunk, understorey). Wet deposition is modelled using interception fractions for each department. Interception fractions are calculated using a modified version of the formula suggested by [Müller, Pröhl 1993].

$$fw = LAI\ .\ c_f\ .\ S_i\ /R\ .\ [1-\exp(-\ln 2\ .\ R\ /pS_i)],$$

where LAI is the leaf area index at time of deposition, c_f the fraction of tree covered area, S_i retention coefficient, R rainfall during deposition event, and p a coefficient characterising the storage capacity of each compartment. For compartments other than crown, c_f is chosen equal one, and p equal to three as in [Müller, Pröhl 1993]. Wet deposition is calculated successively for crown layer, trunk layer, understorey and soil.

I–8.2.2. Forest dynamics

After the deposition event, transfer processes distribute radionuclides between the compartments. The removal and transfer processes are described by differential equations. The processes considered currently are weathering from vegetation, litterfall from trees, foliar absorption, and uptake from soil. In the initial phase after deposition, also runoff is considered. Radionuclides deposited on soil become available to the root uptake with delay. Root uptake is controlled by nuclide specific rates. Bioavailable radionuclide (here: radiocaesium) fraction in soil is reduced by fixation and vertical migration. During ground frost and snow cover most transfer processes are essentially slower than in the growth period.

The radionuclides metabolized through foliar absorption are distributed in crown. The needle year classes exposed directly to deposition are considered in values of weathering rates for the crown in the first four years. Outputs of the forest dynamics submodule to dose calculation routines are the total activities in the compartments crown, trunk, understorey, vegetation and soil, in Bq/m^2. Tree is divided into crown and trunk below the crown for purposes of external dose calculation.

I–8.2.3. Activity concentrations in forest products

For the first ten days after deposition, the activity concentration in mushrooms is related with deposition to understorey and translocation and afterwards using transfer factors from soil.

Until the end of the first year after deposition berries are supposed to be contaminated by translocation of radionuclides from understorey via foliar absorption and for all later times using transfer factors from soil

In the first year after deposition we consider that game animals are contaminated through ingestion of contaminated understorey feed, and afterwards transfer factors from soil are used.

I–8.3. MODEL–DATA COMPARISON

I–8.3.1. Activity concentrations in mushrooms, berries and game meat (IPSN)

IPSN calculated activity concentrations in mushrooms, berries and game using the forest food chain dose calculation module (FDMF) of RODOS.

Some values of transfer coefficients are as follows:

Cantharellus cibarius, Boletus edulis	0.05 m^2.kg^{-1}	(fresh weight)
Xerocomus badius, Suillus luteus	0.5 m^2.kg^{-1}	(fresh weight)
Bilberry	0.03 m^2.kg^{-1}	(fresh weight)
Roe deer	0.1 m^2.kg^{-1}	(fresh weight)

Calculation of foodstuff concentrations received input from the dynamic module of radionuclide transfer in forest ecosystems developed by STUK. This module was still under development and differs somewhat from the (also preliminary) version used by STUK. This intercomparison exercise was an opportunity for us to test our modules and to improve the predictions.

I–8.3.2. Trees and understorey vegetation (STUK)

The components of the crown (needles, branches, and wood and bark of the stem) were considered when estimating parameters for the crown. Stem under the canopy is another tree compartment of FDMF. The distribution of ^{137}Cs in tree after deposition was derived from data for North European pine forests (Mälkönen 1974; Nygrén et. al 1994; Raitio & Rantavaara 1994).

The endpoints calculated for the BIOMASS scenario are concentrations in different tree fractions defined considering the whole tree. They were derived from FDMF normal outputs with a procedure analogous with that used in estimation of input parameters for crown and trunk. Activity concentrations in berries were derived from those of undesrstorey dwarfs.

Input parameters in FDMF were adjusted for the 50 years old pine growing at the site described in the test scenario.

The parameters related to the forest scenario were:
Crown height: 11 m
Trunk height: 11 m
Crown density: 2.01 kg m^{-3}
Trunk density: 1.43 kg m^{-3}
Weathering rate for crown (includes litterfall):
0.016 d^{-1}(first 60 days);
2.89 E-3 d^{-1}(following 10 months);
6.93 E-5 d^{-1}(later)
Weathering rate for trunk:
0.050 d^{-1}(first 60 days);
0.004 d^{-1}(following 10 months);
7.0 E-6 d^{-1}(later)
Runoff rate: 0.006 d^{-1} in April–October, applied until a fraction of 0.03 is reached
Foliar absorption rate: 0.013 d^{-1} (during the first 15 days)
Soil fixation and migration rate: 9.5 E-5 d^{-1}.

References

MUELLER, H., PROEHL G., ECOSYS-87: A dynamic model for assessing radiological consequences of nuclear accidents, Health Phys. **64** (3) (1993) 232–252.

MÄLKÖNEN, E., Annual primary production and nutrient cycling in some Scots pine stands, Comm. Inst. For. Fenn. 84.5, Helsinki (1974).

NYGREN, P., HARI, P., RAUNEMAA, T., KULMALA, M., LUOKKANEN, S., HOLMBERG, M., NIKINMAA, E., Behaviour of ^{137}Cs from Chernobyl fallout in Scots pine canopy in southern Finland, Can. J. For. Res. **24** (1994) 1210–1215.

RAITIO, H., RANTAVAARA, A., Airborne radiocaesium in Scots pine and Norway spruce needles, Sci. Tot. Env. **157** (1994) 171–180.

I–9. FORM (IAEA MODEL)

M.J. Frissel
Heelsum, Netherlands

I–9.1. DESCRIPTION OF IAEA FOREST MODEL W995_1_2 (APPLIED TO SCENARIO 1)

The IAEA forest model has been developed as a decision tool for the evaluation of countermeasures for contaminated forests. It consists of three parts: an ecological part, a dose assessment part and a financial part.

It considers in the first place countermeasures as immediate cutting and postponement of cutting. The dose resulting from harvested wood by industrial or domestic applications is included. The latter dose is the main problem of caesium-contaminated wood.

Furthermore measures are considered as no consumption of forest products, no admittance into forests, fertilisation, etc.

The model covers a period of 100 years. A detailed description of an earlier, simpler, version was described by Frissel et al, 1995.

I–9.1.1. Contributors to the model (alphabetical order)

—	M. Crick (Initiator):	Parameter sensitivity studies
—	M.J. Frissel:	Soils, uptake from soil, wood, computer programming
—	E. Holm:	Dose calculations, industrial applications
—	C. Robinson:	Economic part
—	G. Shaw:	Tree, computer programming

The present version of the model includes the dose assessment part, but not the financial part. The model description is limited to the radioecological part.

I–9.1.2. Description

The structure of some versions of the model is shown in Figure I-11. The model is very easy to adapt to specific needs.

The model contains the following radioecological components:

The *dynamic* compartments, which provide the core of the model. They have the form:

$$C_t = C_{t-1} + R*dt ,$$

where C_t is the current concentration in a compartment and C_{t-1} the concentration in the foregoing year, and R is the transfer rate between two compartments and dt is the time increment.

The *dynamic* compartments are: Cs in wood, bark, litter, organic matter, mineral surface soil, deep soil, fixed Cs and a sink.

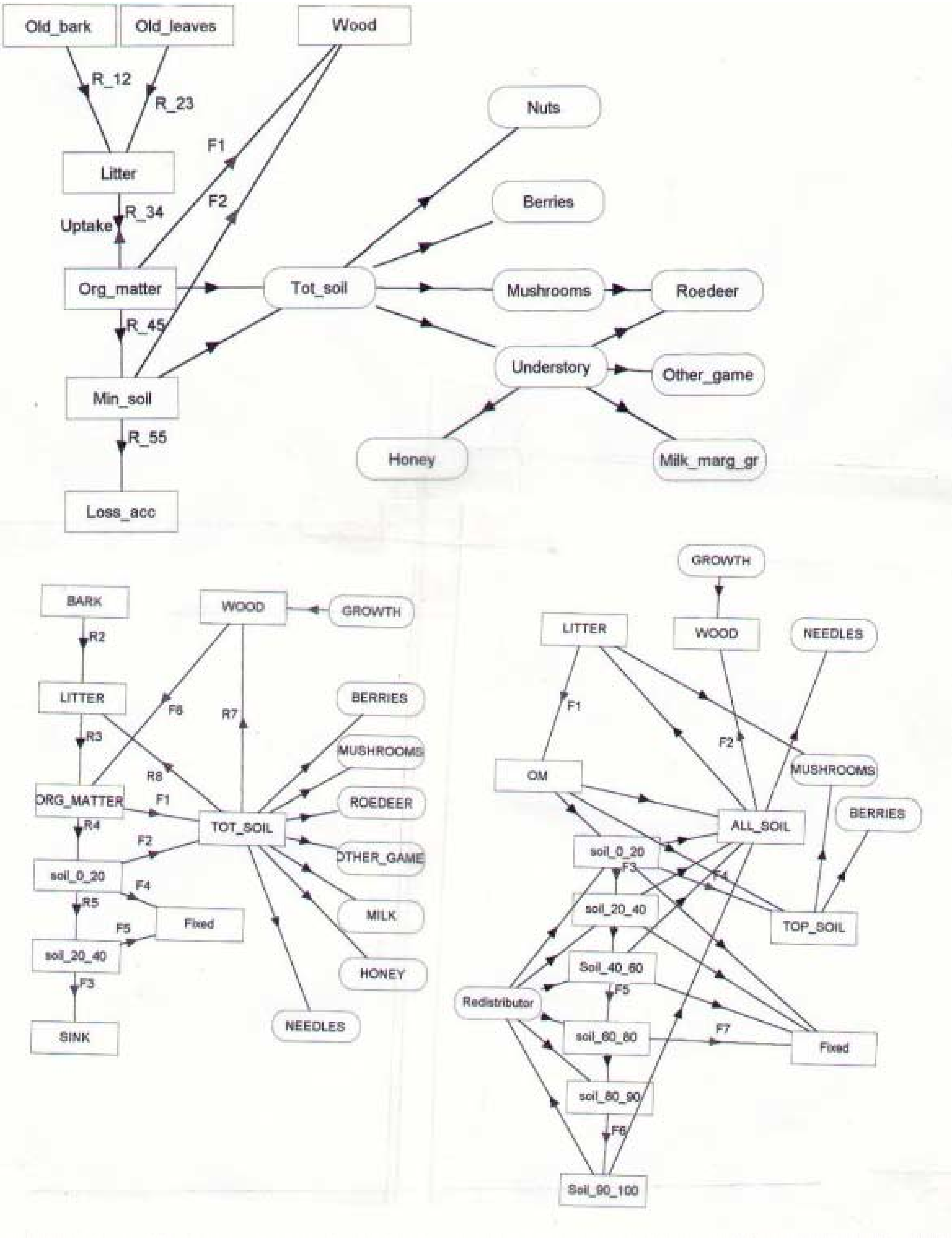


FIG. I-11. The IAEA Forest Model.

Top: The ecological part of the model as developed in 1995.
Bottom left: Model as adapted for Scenario 1 (forest Ukraine).
Bottom right: Model as adapted for Scenario 2 (waste deposit).

The flow of Cs between the compartments is calculated by rate equations. The rate controlling parameters are the transfer of Cs from: Bark-to-litter, litter-to-OM, OM-to-surface soil, surface soil-to-deep soil, deep soil-to-sink and mineral soil-to-fixed Cs. The surface mineral soil layer covers the 0–20 cm layer, the deep layer the 20–40 cm layer. The deep layer is added to have a possibility to model uptake from this layer. The sink or loss-accumulator is added to allow a material balance for control purposes.

The uptake rate of Cs by wood is annually calculated for the annual wood growth increment, from the Cs pool in the OM and mineral surface soil layer and the use of a transfer factor. This quantity is added to the Cs which is already present in the wood from foregoing years. The production of wood is a function of the age of the tree, a typical example of a weight calculation is:

$$\text{IF (age<60, G = 0.0375 * age}^{2.12}\text{, G = 220 + (age–60) * 11.9)}$$

The key equation for the concentration of Cs in wood is:

$$Cs_{(wood,t)} = (Cs_{(OM+min\ surf\ soil)} * TF * dt * dG + Cs_{(wood,\ t-dt)} * G_{(t-dt)})/G_t$$

where

$Cs_{(wood,t)}$	is the average Cs concentration in wood at time t, Bq/kg;
$Cs_{(OM+min\ surf\ soil)}$	is the Cs in OM and surface mineral soil layer at time t, Bq/m^2;
TF	is the transfer factor, m^2/kg;
dt	is the time increment, usually 1 year;
dG	is the growth of wood during time increment, kg;
$Cs_{(wood,\ t-dt)}$	is the average Cs in wood at time (t–dt), Bq/kg;
$G_{(t-dt)}$	is the weight of wood at time (t–dt), kg; and
G_t	is the weight of wood at time t, kg.

Because of the free choice of time of cutting and consideration of the specific growth rate, the wood contamination programming is rather complicated.

The translocation of Cs between annual wood increments is not considered. There is a parameter included which accounts for Cs loss from wood to OM, this process is called biodecay.

There is no compartment for leaves/needles or newly formed bark. For assessment calculations these compartments are not relevant. Instead, the amounts taken up by leaves, needles and new bark are immediately added to the litter pool. With a time step of one year, this is for deciduous forests very correct, for coniferous forests it leads to a conservative estimate of the dose.

To allow a comparison with other models, the concentration in leaves/needles is calculated in the same way as the concentration in other forest products. This is clearly shown in the flowchart. The initial contamination of the leaves is not considered.

The bark compartment is only used to store Cs upon the initial contamination. Transfer is to litter only, transport from bark to wood is not included.

Because the litter production is expressed as Bq/kg, and the litter compartment accounts for the total Cs in the layer, the litter yield is an input parameter.

Cs fixation (i.e. irreversible absorption in soil) is considered by a first order rate process.

The model includes the following *non-dynamic* compartments: Mushrooms, berries, honey, nuts, milk, game, roedeer, grass and leaves/needles. The Cs concentrations in these compartments are calculated from the amount of Cs present in OM + mineral surface soil layer by multiplication by a simple aggregated transfer factor. For mushrooms also the litter compartment is considered.

Because there is no canopy compartment the initial distribution of a radioactive contamination is for the greater part assumed to be present in the litter layer, smaller fractions are allocated to bark, OM and mineral surface soil.

I–9.2. PARAMETER SELECTION

Most parameters were selected during the development of the model by Shaw and Frissel in 1995 and are often based on IAEA Handbook on Transfer parameters (TR 364). The soil to plant parameters for wood, leaves/needles, grass and berries are based on generic soil system reference values (Table I-3). These values have to be multiplied by conversion factors for specific crop groups (Table I-4, Frissel 1999).

The site description is not clear as far as the nutrient status is concerned. From a comparison of calculations and observed data it appears that the expected parameter values for soils with a low nutrient status show the best fit.

I–9.3. ADAPTATIONS FOR VERSION W995_2_1 (SCENARIO 3)

To adapt the IAEA model to the requirements of the waste deposit scenario two modifications were introduced.

I–9.3.1. The redistributor

The main water flow is downwards, even capillary rise will therefore not result in an upward flow of material. Diffusion is extremely slow and therefore not relevant.

The main mechanism that causes an upward transport of radionuclides in a soil is mechanical mixing due to the presence of animals. Mechanical mixing may be much more important in soil then is generally recognized, Long term studies show that the migration of Pu and Am in the upper soil layers (say upper 50 cm) is almost equal to the migration of Cs and Sr. Yet the solubility of Sr and Cs is much larger than of Pu and Am. It has been assumed that the mixing is caused soil animals.

TABLE I-3. EXPECTED REFERENCE TRANSFER FACTORS OF Cs. REFERENCE VALUES EXPRESSED AS (Bq/kg DRY CROP)/(Bq/kg SOIL IN THE UPPER 20 cm OF SOIL)

Nutrient status	Soil type	Reference tf's of Cs for accidental releases	
		Expected value	Range
high nutrient status, pH >4.8	all soils	0.05	0.02–0.1
medium nutrient status, pH >4.8	Clay and loam soils	0.1	0.05–0.5
	Sand, peat and other soils	0.2	0.1–0.5
	Clay soils	0.5	0.2–1
	Sand and other soils	0.7	0.2–2
	pH >4.8	0.7	0.2–2
low nutrient status OR pH <4.8	Peat soils pH <4.8 Normal moisture	1.4	0.4–4
	Wet, gley	8	2–20
	Soils with exchangeable K <0.05 meq/100 g	5	2–10

TABLE I-4. RECOMMENDED CONVERSION FACTORS

Recommended conversion factor	Cereals	Grass	Fruit*	Leaves of woody species as tea and thyme**
Cs	1	4.5	5	20
Standard deviation for Cs	11	4		

* Values for berries may be higher

**Also applied to needles and newly formed wood.

The redistributor assumes that a certain fraction (typical value 0.03 per year) of the activity in the 90–100 cm layer is distributed over the other layers. (typical fraction values for the upper 10 cm layers 0.2, for the layer 80–90 0.84). Application of these values gives a distribution which is more or less in agreement with the observed data as provided in the scenario.

I–9.3.2. Uptake from various layers

The uptake is assumed to depend on rooting depth of the tree, which in turn is a function of the age of the tree. The various uptake fractions are listed in Table I-5. The choice is arbitrarily and not supported by any reported observation.

TABLE I–5. UPTAKE FRACTIONS AS A FUNCTION OF AGE AND SOIL DEPTH

	Soil depth					
Period y	0–20 cm	20–40 cm	40–60 cm	60–80 cm	80–90 cm	90–100 cm
0–10	1	0	0	0	0	0
11–20	0.5	0.5	0	0	0	0
21–100	0.3	0.3	0.05	0.05	0.01	0.01

References

FRISSEL, M.J., SHAW, G., ROBISON, C., HOLM, E., CRICK, M., Model for the long term countermeasures in forests, in “Radioecology and the restoration of radioactive-contaminated sites (F.F. Luykx and M.J. Frissel, Eds) Kluwer Academic Publishers (1995).

FRISSEL, M.J., Updating soil to plant transfer factors of Cs and Sr, available as Internal Working Document (1999).

I–10. FORWASTE

A.V. Konoplev, A.A. Bulgakov
Institute of Experimental Meteorology,
SPA “Typhoon”,
Obninsk, Russian Federation

I–10.1. THE MODEL STRUCTURE

A conceptual scheme of the model is given on Figure I-12. The model comprises two blocks – TREE and SOIL. The block SOIL consists of compartments corresponding to successive layers of soil. Number and thickness of soil layers depend on modelling objective and availability of input soil characteristics. Thickness and weight of the upper layer increase with time as a result of leaves (needles) fallout. The block TREE consists of compartments corresponding to different parts of tree. Tree parts characterized by biomass–age functions. Active roots biomass is given separately for each soil layer. For a given modelling problem some compartments can be combined. For example, radionuclide exchange between TRUNK and EXTERNAL WOOD in a young tree is quite fast and there is no reason to consider these two compartments separately. From the other hand, some part of tree can be presented in the model in more details. In particular, the compartment NEEDLES can be divided into several compartments according to the age.

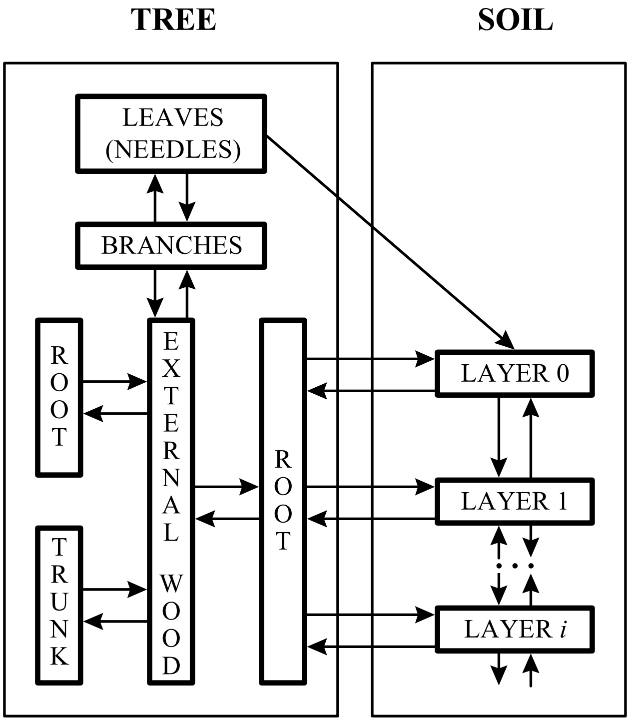


FIG. I-12. The model scheme.

I–10.2. RADIONUCLIDE FLUXES BETWEEN COMPARTMENTS

Uptake of radionuclides by tree roots is described in existing mathematical models as an irreversible process [1]. However, it was shown that radiocaesium introduced in the tree trunk is partially transferred to the root system and, further, to the soil around roots [2]. This means that trees uptake radiocaesium reversibly and able to release it to soil. Data on ^{137}Cs content in pine roots at different depths indicates that radionuclide downward transport via roots can be an important mechanism of redistribution of radioactive contamination in soil. In the Chernobyl NPP vicinity activity concentration of ^{137}Cs in pine roots in the lower soil layers is almost the same as in the upper one, in spite of several order of magnitude difference in soil contamination [7,8]. Based on these observations we describe root uptake of radionuclides as a reversible process and consider tree root system as one compartment.

Radiocaesium transfer from the root exchange complex to the underground part of plants is relatively fast. Characteristic time of this process for grassy plants is about 1 day [5]. The model considers the root uptake rate to be limited by the rate of radionuclide diffusion and convection through the soil layer adjacent to roots:

$$F_{i,w} = \left(\frac{D_E}{l_+} + V_T \right) \rho_i R_i B_R \sigma \left(\frac{S_i}{m_i} - \frac{W}{CR_i B_W} \right) \tag{1}$$

where:

D_E is the radionuclide effective diffusion coefficient in soil, m^2/s;
V_T is the velocity of radionuclide transport with the water taken up by roots, m/s;
ρ_i is the density of the i–th layer of soil, kg/m^3;
R_i is the fraction of active roots in the i–th layer of soil;
σ is the specific surface of active roots, m^2/kg;
B_R, B_W and m_i are dry masses of active roots, wood and the i–th layer of soil respectively, kg/m^2;
W and S_i are the radionuclide content in wood and in the i–th layer of soil respectively, Bq/m^2;
CR_i is the ratio of the radionuclide concentration in wood to that in the i–th layer of soil when the flux from the layer to wood is equal to that from wood to the layer;
l_+ is the thickness of the soil layer which is depleted in radionuclide as a result of its transfer from soil to roots, m.

The reverse flux from tree to soil occurs by diffusion only:

$$F_{w,i} = \frac{D_E}{l_-} \rho_i R_i B_R \sigma \left(\frac{S_i CR_i}{m_i} - \frac{W}{B_W} \right) \tag{2}$$

where:

l_- is the thickness of the soil layer which is enriched in radionuclide as a result of its transfer from roots to soil, m.

Values of l_+ and l_- are, generally, different and time-dependent. Then the effective flux is directed from soil to tree, almost equilibrium activity concentration in tree can be reached as a result of radionuclide transfer from a relatively thin layer of soil adjacent to roots. Then the reverse flux prevails, equilibrium can be established only after spreading of released radionuclide through the whole root zone volume. Therefore, l_+ is normally smaller than l_-.

Fluxes within tree are assumed to be reversible and described by first order kinetic equations. Relative concentrations of ^{137}Cs in different tree parts are approximately constant for a given species all over the Chernobyl zone [4, 6, 7]. This indicates that radiocaesium transfer within tree is quite fast and activity distribution between tree parts is close to equilibrium. The only exception is, probably, radionuclide exchange between external wood and trunk for a tree after a certain age.

Radionuclide flux from tree crown to the soil surface (F_{LS}, Bq/m^{2}y) is calculated as follows:

$$F_{LS} = (\phi f_L + r I_R)L/B_L \tag{3}$$

where

ϕ	is the ratio of radionuclide concentration in litterfall to that in leaves (needles) as a whole;
f_L	is the litterfall rate, kg/m^{2}y;
L	is the radionuclide content in leaves (needles), Bq/m^2;
B_L	is the biomass of leaves (needles), kg/m^2;
r	is the ratio of radionuclide concentration in the crown water to that in leaves (needles), kg/L;
I_R	is the average precipitation rate, L/m^{2}y = mm/y.

Radionuclide fluxes between soil layers are described by first order kinetic equations. Rate constants are estimated, if possible, from experimental data on radionuclide vertical distribution in soil. Given no experimental data, rate constants are calculated as a function of effective diffusion coefficient and convective transport velocity. Effective diffusion coefficient and convective transport velocity are, in turn, estimated using all available information about meteorological conditions, soil properties and radionuclide speciation.

The main distinctions of this model from existing models are:

- root uptake of radionuclide is considered to be a reversible process;
- root system depth distribution are given as a function of time;
- the model allows, from the one hand, to estimate the tree role in radionuclide vertical migration in soil, and, from the other hand, to predict radioactive contamination of tree as a function of radionuclide vertical distribution in soil.
- Model calculations shows that radiocaesium transfer from the surface radioactive waste disposal site via root system of pine can lead to significant contamination of the soil surface layer.

References

[1] AVILA R., MOBERG L., HUBBARD L., Modelling of radionuclide migration in forest ecosystems, a literature review, Swedish Radiation Protection Institute Report 98:07, Stockholm, SSI, 36 p (1988).

[2] OLSON, J.S., Distribution of roots and radiocaesium migration in them in a mesophilic forest in Tennessee, In: Methods of investigation of productivity of root systems and organs of rhizosphere, Leningrad, Nauka, 143–149 (1968) (in Russian).

[3] SAK M.M., Accumulation of radionuclides by pine roots in forests of different types, In: Abstracts of Radiobiological Congress, Kiev, September 1993, Puschino, p. 880 (1993) (in Russian).

[4] SCHEGLOV, A.I., Biochemistry of technogenic radionuclides in forest ecosystems, Moscow, Nauka, 268 p. (1999) (in Russian).

[5] SOKOLIK, A.I., DEMKO, G.G., GOROBCHENKO, N.E., YURIN, V.M., Main mechanisms of ^{137}Cs root uptake, Radiation Biology, Radioecology, **37** 5, (1997) 787–795 (in Russian).

[6] IPATEV V.A., BRAGINSKY, V.F., BULAVIK, I.M., et al., “Forest. Man. Chernobyl”, Gomel, 454 p (1999) (In Russian).

[7] DVORNIK, A., ZHUCHENKO, T., Model FORESTLIFE and prediction of radioactive contamination of forests in Belarus, In: Linkov, I., Schell, W.R. (Eds), Contaminated forests, NATO Science Series **58**, Kluwer Academic Publishers, Dordrecht, The Netherlands (1999) 185–194.

I–11. LOGNAT

M. Scimone
Department of Biology,
University of Trieste,
Trieste, Italy

LOGNAT is written in software STELLA II and is a simple compartment-type model assessing the transfer of ^{137}Cs from an initial event (deposition at t=0) in function of time (years). The variable state is indicated in Bq m^{-2}. Main assumptions o the model are:

(1) The model calculates the circulation of ^{137}Cs in a closed system (forest), assuming an initial deposition in the litter, holorganic and leaves compartment as input values.

(2) No losses (sink) from the system are accounted for.

(3) Corrections are considered for decay, assuming ^{137}Cs half-life of 30 yrs (k = 0.023).

(4) Transfer between compartments are expressed as first order kinetics, in form of dimensionless parameters (fraction of total amount, 1/yr).

(5) Transfer parameters have been derived from experimental data (litter decomposition, soil and leaves sampling, etc.) and literature.

(6) The forest biomass evolution at long-term scale (i.e. 50 years) during simulations is calculated as growth rate function of the standing biomass.

(7) Uptake rates are considered in function of the standing biomass and productivity. Uptake rate is expressed as a fraction parameter per standing biomass unit (1/ (yr * kg of biomass)).

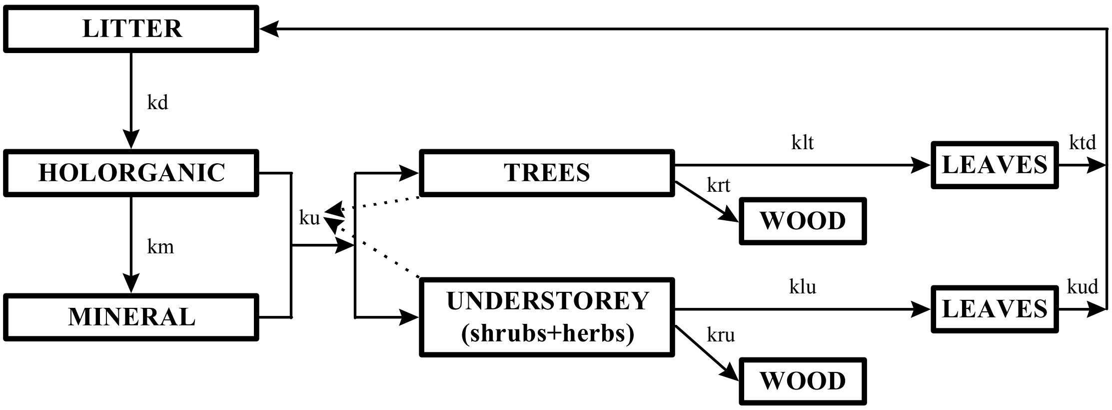


FIG. I-13. Model LOGNAT structure.

I–12. FORSUN

A.V. Konoplev, A.A. Bulgakov
Centre for Environmental Chemistry,
SPA “Typhoon”,
Obninsk, Russian Federation

I–12.1. OBJECTIVES

The main objectives of the model are:

- to investigate mechanisms of radiocaesium uptake by plants, in particular possible role of fungi as a mediator in soil–plant transfer;
- to describe qualitatively and (if possible) quantitatively radiocaesium distribution in soil–plant system as a function of time;
- to predict radiocaesium concentration in forest plants and mushrooms using minimum of parameters for which the only way of estimation is the model calibration.

I–12.2. THE MODEL STRUCTURE

The model compartments and processes of radiocaesium transfer between compartments are given in Figure I-14. The model is constructed with the software package STELLA 5.0. For a more detailed description of the system some compartments can be divided into several parts. For example, the compartment ”Plant” can be represented as a combination of Root Exchange Complex (REC), Root, and Aboveground part. For more accurate modelling of radiocaesium vertical transport and for taking into account changes in soil properties with depth, compartments Soil 1 and Soil 2 can be divided into several sub-compartments (layers). The optimal level of model complexity depends on the modelling objectives and availability of information about soil, plant and mushroom characteristics. The most realistic approach is using soil horizons as elements of the soil block of the model.

I–12.3. MECHANISMS AND RELATIVE SIGNIFICANCE OF INTERCOMPARTMENT EXCHANGE PROCESSES

I–12.3.1. Plant/Mycelium

Given high radiocaesium concentrations in fungi, the large surface area of mycelium and its connection with roots via mycorrhizae, many researchers have speculated that in forest soils mycelium is the main source of radiocaesium in plants. However, no direct evidence of radiocaesium exchange in the plant/mycelium system has been reported. Moreover, the following indirect data indicate that there is no significant caesium exchange between plant and mycelium:

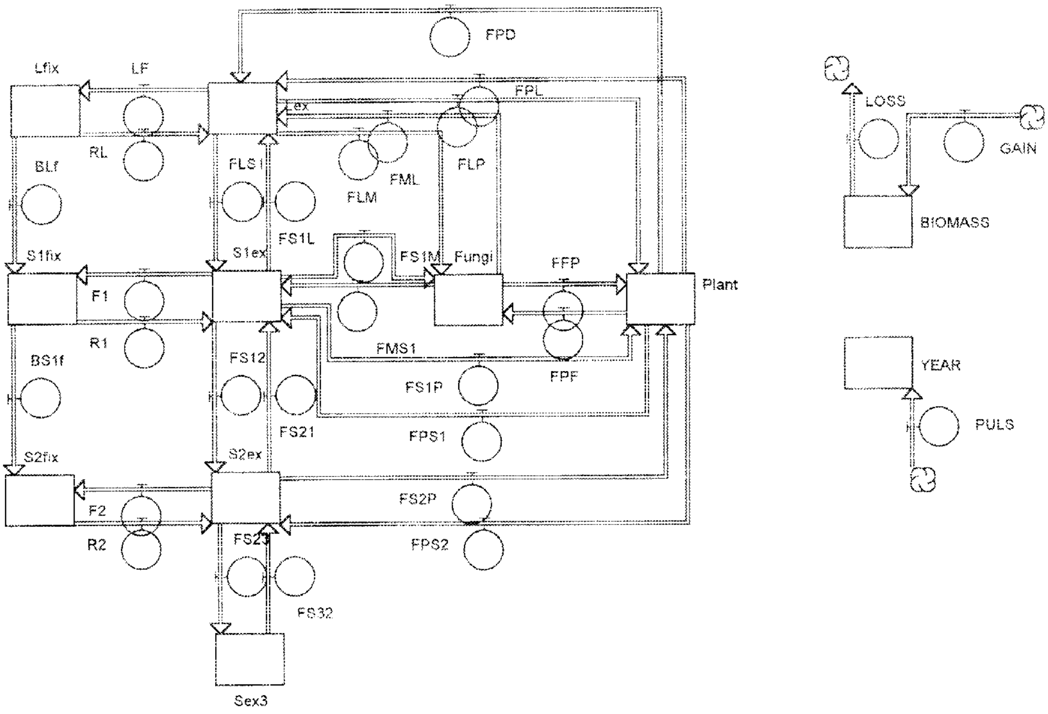


FIG. I–14. Conceptual scheme of dynamic model of radionuclide soil-fungi and soil-plant transfer with consideration of role of fungi as a mediator in soil-plant transfer.

- Radiocaesium dynamics in fungi and at least in some plant species in 1987–1990 are different (Shutov et al., 1998). In all mushroom species in Bryansk region the concentration of Chernobyl radiocaesium increased during the first 4–5 years after deposition while no increase or even decrease of radiocaesium concentrations in plants were observed.
- In meadow soils with mycelium radiocaesium CR is lower then in meadow soils without mycelium (Clint et al., 1991; Dighton et al., 1991). Authors believe that this is a result of caesium immobilisation in mycelium. The same result was obtained with plants grown in nutrient solution with and without mycorrhizae (Clint and Dighton, 1992).
- There are some differences in reported values of $^{137}Cs/^{134}Cs$ and $^{137}Cs/^{133}Cs$ ratios for fungi and plants (Yoshida and Maramatsu, 1998), although statistical significance of this difference is questionable.
- Recently it was shown that radiocaesium concentration in mycorrhized heather is up to 40% higher than that in non-mycorrhized heather (Strandberg and Johansson, 1998). The increased ^{134}Cs uptake from inoculated soil was observed for three variants of soil treatments, while for the fourth inoculation had no effect on the radionuclide concentration in the plant.

Based on these observations, it is impossible to make quantitative and even qualitative conclusions about radiocaesium exchange in plant/mycelium system. One of the objectives of our model is to investigate possible effect of this exchange on radiocaesium concentration in plants and mushrooms as functions of time.

I–12.3.2. Plant/Litter

Litter – Plant

Some researchers believe that radiocaesium uptake by plants from the litter is more intensive than from the soil (Firsakova et al., 1980). No direct study of relative importance of radiocaesium uptake by plants from litter and soil were reported. In the first version of the model we assume that the plant can uptake radiocaesium directly from the litter.

Plant – Litter

Radiocaesium uptaken by plants is incorporated in the litter (in water-soluble and exchangeable forms, (SPA “Typhoon” 1986, unpublished data) at the end of the growing season.

I–12.3.3. Plant/Soil

Radiocaesium enters to plant cell from soil solution through potassium channels (selective pathway) and through non-selective ionic leakage (Sokolik et al., 1997).

I–12.3.4. Mycelium/Litter

Litter – Mycelium

Increase of ^{137}Cs/^{134}Cs ratio in fungi in the first years after deposition proves that mycelium uptakes radiocaesium from the soil better than from the litter, but relatively high concentration of ^{134}Cs in fungi in 1986 indicates that some uptake from litter also take place.

Mycelium – Litter (see Mycelium – Soil)

I–12.3.5. Mycelium/Soil

Soil – Mycelium

It is well known that mushrooms can accumulate significant amount of radiocaesium from contaminated soils. Some researchers believe that radiocaesium concentration in mycelium is approximately the same as in mushroom fruitbodies (Olsen, 1990; Nikolova et al., 1997). There are some evidences that in respect to uptake by mushrooms caesium is a very close analogue of potassium (Olsen, 1990; Perkinns and Gadd, 1993).

Mycelium – Soil

Radiocaesium transfer from mycelium to soil occurs as a result of mycelium dying and, probably, due to ionic leakage.

I–12.3.6. Processes in soil

Two types of processes lead to radiocaesium redistribution and chemical speciation transformation in soils:

- Biological processes include litter decay and bioturbation. Expert judgements based on experimental data are used for description of these processes.

— Physico-chemical processes include diffusion, convection, adsorption (cationic exchange), fixation and remobilization. Several models and methods of parameters values estimation (theoretical and experimental) were developed during the last decade for description of these processes (Cremers et al., 1988; Konoplev and Golubenkov, 1991; Konoplev et al., 1992; Bulgakov and Konoplev, 1996; Konoplev and Konopleva 1999; Konoplev and Bulgakov, 1997). Here we use the Konoplev and Golubenkov (1991) model (Konoplev and Golubenkov, 1991) for description of radiocaesium fixation and vertical migration in the soil, which was validated in frame of BIOMOVS II.

I–12.4. INTERCOMPARTMENT FLOWS DESCRIPTION

All included processes are assumed to be reversible and described with the first order kinetic equations. The rate constants values are calculated from the following set of equations:

$$\begin{aligned} k_{ij} + k_{ji} &= \tau_{ij}^{-1} \\ k_{ji} / k_{ij} &= CR_{ij} \end{aligned} \tag{1}$$

where:

k_{ij} and k_{ji} are first order rate constants for radiocaesium transfer from compartment i to compartment j and from compartment j to compartment i respectively;

τ_{ij} is characteristic time of radiocaesium transfer from compartment i into compartment j;

CR_{ij} is concentration ratio equal to the ratio of radiocaesium concentration in compartment i to that in compartment j in (Bq/kg d.w.)/(Bq/kg d.w.);

The flow from compartment i into compartment j (F_{ij}, Bq.y^{-1}.m^{-2}) is calculated as follows:

$$F_{ij} = \tau_{ij}^{-1}\left(1 + CR_{ij}\right)^{-1}\left(Q_i - Q_j\, m_j\, m_j^{-1}\, CR_{ij}\right) \tag{2}$$

where:

Q_i and Q_j are activity of compartments i and j correspondingly in Bq.m^{-2};

m_i and m_j are mass of compartments i and j correspondingly (or time dependent biomass of biological compartments), kg.m^{-2}.

I–12.5. ESTIMATION OF PARAMETER VALUES

I–12.5.1. Soil and litter compartments

Parameter estimation methods for the soil block of the model are given in (Konoplev and Golubenkov, 1991).

I–12.5.2. Biological compartments

I-12.5.2.1. Time scale of exchange in Soil/ Plant(mycelium) system

The time scale of radiocaesium exchange in the plant/nutrition solution system, measured by Sokolik et al. (1997) is approximately equal to 1 day. In soils the intensity of radiocaesium uptake by plant roots depends on the rate of its transfer from bulk soil solution to the solution in the near root layer and on the rate of the radionuclide exchange in the plant/soil solution system. In soils poor of potassium, radiocaesium and potassium concentration in the nearest vicinity of the root is usually significantly smaller than in the bulk soil solution (Nye and

Tinker, 1980; Smolders et al., 1996). In these soils the rate of caesium cations uptake by plant is controlled by their diffusional and convective transport to the root surface.

In soils with potassium concentration in the pore solution about 1 mM there is almost no dependence of dissolved radiocaesium and potassium content on the distance from the root surface (Nye and Tinker 1980; Smolders et al., 1996). It means that radiocaesium transport to the near root zone from the bulk soil is not significantly slower than its absorption by roots. In our model it is assumed that time scale of radiocaesium transfer from forest soil into plant (mycelium) is somewhat longer than from nutrition solution and equal to 1 week.

I-12.5.2.2. Soil-to-plant concentration ratio

According to (Konoplev et al., 1998), soil-to-plant concentration ratio for soils rich of potassium can be calculated as follows:

$$CR = B \times A \tag{3}$$

where:

B is specie dependent constant; and
A is the bioavailability factor.

Bioavailability factor A is assumed to be proportional to the fraction of radiocaesium in the root exchange complex and can be calculated as follows (Konoplev et al., 1998):

$$A = \frac{\alpha_{ex} PNAR}{RIP^{ex}} \tag{4}$$

where:

α_{ex} is the fraction of the exchangeable ^{137}Cs in the root zone;
PNAR is the potassium–ammonium adsorption ratio; and
RIP^{ex} is the exchangeable radiocaesium interception potential.

$$PNAR = \frac{[K]_w + K_c(NH_4/K)[NH_4]}{\sqrt{[Ca]+[Mg]}} \tag{5}$$

where:

$[K]_w$, $[NH_4]_w$, $[Ca]_w$, $[Mg]_w$ are potassium, ammonium, calcium and magnesium concentrations in the soil solution in $mM.l^{-1}$, correspondingly;
$K_c(NH_4/K)$ is the selectivity coefficient of potassium–ammonium cation exchange on selective adsorption sites (FES).

RIP^{ex} values are available only for few soils. For other soils the following simplified equation for A was developed (Progress Report on RYS-6.15B, 1998; Konoplev et al., 1998):

$$A^* = \frac{10\alpha_{ex}}{K_c^{eff}(Cs/K)[Ca]_{ex}} \tag{6}$$

where:

$K_c^{eff}(Cs/K)$ is the effective selectivity coefficient of caesium–potassium cation exchange;
$[Ca]_{ex}$, $[Mg]_{ex}$ and $[K]_{ex}$ are exchangeable calcium, magnesium and potassium concentrations in soil in $meq.kg^{-1}$, correspondingly.

When properties of the root zone change significantly with depth, the effective availability factor value is calculated as a sum of radiocaesium fractions in the root exchange complex of roots located in sequential soil layers:

$$CR = B\frac{\sum_i \left(A_i \times R_i \times Q_i\right)}{\sum_i Q_i} \tag{7}$$

where:

A_i is availability factor for soil layer i;
Q_i is radiocaesium activity in soil layer i in Bq.m^{-2}; and
R_i is portion of total root exchange complex in soil layer i.

I-12.5.2.3. Soil-to-fungi concentration ratio

Unfortunately, no estimation method for radiocaesium concentration ratio in mushroom as a function of specie and soil properties was reported in literature. There are some evidences that in respect of uptake by mushrooms from nutrient solutions caesium is a very close analogue of potassium (Olsen, 1990; Perkinns and Gadd, 1993). These observations allow us to assume that radiocaesium/potassium ratio in mushroom is directly proportional to that in soil solution and discrimination factor is close to unity:

$$\frac{\left[{}^{137}Cs\right]_M}{\left[K\right]_M} = \mu\frac{\left[{}^{137}Cs\right]_w}{\left[K\right]_w} \tag{8}$$

where:

$[{}^{137}Cs]_M$ and $[K]_M$ are radiocaesium and potassium concentrations in mushrooms (mycelium) in Bq.kg^{-1} and mM correspondingly;
$[{}^{137}Cs]_w$ and $[K]_w$ are radiocaesium and potassium concentrations in soil solution in mycelium location layer; μ is discrimination factor, $\mu \approx 1$.

Using equation relating $[{}^{137}Cs]_w$ with $[{}^{137}Cs]_{ex}$ via RIP^{ex}, we get:

$$\frac{\left[{}^{137}Cs\right]_M}{\left[{}^{137}Cs\right]_{ex}} = \mu\frac{\left[K\right]_M}{RIP^{ex}} \tag{9}$$

Finally, for CR_M we have:

$$CR_M = \alpha_{ex}\mu\frac{\left[K\right]_M}{RIP^{ex}} \tag{10}$$

For soils with unknown RIP^{ex} the following equation can be used:

$$CR_M = \alpha_{ex}\mu\frac{\left[K\right]_M}{K_c^{eff}\left[K\right]_{ex}} \tag{11}$$

So, for mushrooms availability factor $A_M = \alpha_{ex}/RIP \approx \alpha_{ex}K_c[K]_{ex}$, and proportionality coefficient B_M is equal to $\mu[K]_M \approx [K]_M$.

When properties of the mycelium location zone change significantly with depth, the effective availability factor value is calculated as for plants:

$$CR_M = B\frac{\sum_i \left(A_M^i \times M_i \times Q_i\right)}{\sum_i Q_i} \qquad (12)$$

where:

$A_M{}^i$ is availability factor for mushroom/mycelium of soil layer i;
Q_i is activity of soil layer i in $\mathrm{Bq.m^{-2}}$;
M_i is ratio of mycelium content (in $\mathrm{kg.kg^{-1}}$) in soil layer i to that in the mycelium location zone as a whole.

References

APARIN, B.F., ZABOEVA, I.V., LIPKINA, G.C., et al., Podzolic soils of Central and Eastern parts of the European territory of USSR, Moscow, Nauka, 200 p (1981) (In Russian).

AVILA, R., J. Environ. Radioact. (1998) (In press).

BLAMEY, F.P.C., EDMEADES, D.C., WHEELER, D.M., Role of root cation-exchange capacity in differential aluminium tolerance of Lotus species, Journal of Plant Nutrition **13** No. 6 (1990) 729–744.

BULGAKOV, A.A., KONOPLEV, A.V., Diffusional Modelling of Radiocesium Fixation by Soils, Radiation Protection Dosimetry, No. 1/2 (1996) 11–13.

BULGAKOV, A.A., KONOPLEV. A.V., SHKURATOVA, I.G., Empirical time dependencies of ^{137}Cs fraction in the upper layer of soil in the Chernobyl exclusion zone, Russian Soil Sci. (1998) (submitted in Russian).

CLINT, G.M., DIGHTON, J., REES, S., Influx of ^{137}Cs into hyphae of basidiomycete fungi, Mycological Research **95** (1991) 1047–1051.

CLINT, G.M., DIGHTON J., Uptake and accumulation of radiocaesium by mycorrhizal and non-mycorrhizal heather plants, New Phytology **121** (1992) 555–561.

CREMERS, A., ELSEN, A., DE PRETER, P., MAES A., Quantitative Analysis of Radiocaesium Retention in Soils, Nature **335** (1988) 247–249.

CROOKE, W.M., The measurement of the cation-exchange capacity of plant roots, Plant and Soil, **XXI** No. 1 (1964) 43–49.

DIGHTON, J., CLINT, G.M., POSKITT, J., Uptake and accumulation of radiocaesium by upland grassland soil fungi: a potential pool of Cs immobilisation, Mycological Research **95** (1991) 1052–1056.

FIRSAKOVA, S.K., GREBENSCHIKOVA, I.V., Uptake of ^{90}Sr and ^{137}Cs by grassland vegetation, Doklady VASKhNIL **9** (1980) 19–22 (in Russian).

INTERNATIONAL ATOMIC ENERGEY AGENCY, Validation of models using Chernobyl fallout data from southern Finland, IAEA-TECDOC-904, IAEA, Vienna, 483 p (1996).

KONOPLEV, A.V., AVILA, R., BULGAKOV, A.A., DRISSNER, J., JOHANSON, K.-J., KLEMT, E., KONOPLEVA, I.V., MILLER, R., NIKOLOVA, I., POPOV, V.E., ZIBOLD, G., Modelling radiocaesium bioavailability in forest soils, Paper presented on NATO-ARW "Forest Radioecology", Kiev, June 1998 (1998).

KONOPLEV, A.V., KONOPLEVA, I.V., Determination of radiocaesium selective sorption characteristics of soils and bottom sediments, Russian Geochemistry (1999) (in press).

KONOPLEV, A.V., GOLUBENKOV, A.V., Modelling of radionuclides vertical migration in soils after the nuclear accident, Soviet Meteorology and Hydrology **10** (1991) 62–68.

KONOPLEV, A.V., BULGAKOV, A.A., POPOV, V.E., BOBOVNIKOVA, TS.I., Behaviour of Long-Lived Chernobyl Radionuclides in a Soil-Water System. Analyst **117** (1992) 1041–1047.

NIKOLOVA, I., JOHANSON, K.-J., DAHLBERG, A., Radiocaesium in fruitbodies and mycorrhizae in ectomycorrhizal fungi, J. Environ. Radioactivity **17** No. 1 (1997) 115–125.

NYE, P.H., TINKER, P.B., Solute transport in the soil–root system, Moscow, Kolos (1980) (Russian Translation).

Olsen, R.A., Joner, E., Bakken, L.R., Soil fungi and fate of radiocaesium in soil ecosystems, In: Desmet G., Nassimbeni P., Belli M. Proc. of the workshop “The transfer of radionuclides inn natural and semi-natural environments”, Elsevier, London-New York (1990) 657–663.

PERKINS, J., GARD, G.M., Caesium toxicity, accumulation and intracellular localisation inn yeats, Mycological Research **97** (1993) 717–724.

RANTAVAARA, A.H., Transfer of radiocesium through natural ecosystems to foodstuffs of terrestrial origin in Finland, In: Desmet G., Nassimbeni P., Belli M. Proc. of the workshop “The transfer of radionuclides in natural and semi-natural environments”, Elsevier, London-New York (1990) 202–209.

RUHM, W., KAMMERER, L., HIERSCHE, L., WIRTH, E., The ^{137}Cs/^{134}Cs ratio inn fungi as an indicator of the major mycelium location inn forest soil, J. Environ. Radoactivity **35** (1997) 129–148.

SHUTOV, V.N., BRUK, G.J., KADUKA, M.V., BALONOV, M.I., BASALAEVA, L.N., TRAVNIKOVA, I. Dynamics of ^{137}Cs transfer factors from soil into forest mushrooms and berries after the Chernobyl accident, paper presented on IUR Topical Meeting, Mol, June 1998 (1998).

SMOLDERS, E., KIEBOONS, L., BUYSSE, J., MERCKX, R., ^{137}Cs uptake in spring wheat (Triticum aestivum L., ev. Tonic) at varying K supply, II: A potted soil experiment, Plant and Soil **181** (1996) 211–220.

SOKOLIK, A.I., DEMKO, G.G., GOROBCHENKO, N.E., YURIN, V.M,. Basic mechanisms of ^{137}Cs uptake by the root system of plants, Radiation Biology. Radioecology **37** (1997) 787–795 (in Russian).

STRANDBERG, M., JOHANSSON, M., ^{134}Cs in heather seed plants grown with and without mycorrhiza, J. Environ. Radoactivity **40** (1998) 174–184.

YOSHIDA, S., MURAMATSU, Y., Concentrations of alkali and alkiline earth elements in mushrooms and plants collected in a Japanese pine forest, and their relationship with ^{137}Cs, J. Environ. Radoactivity **41** (1998) 183–205.

ANNEX II
HYPOTHETICAL SCENARIO FOR MODEL–MODEL INTERCOMPARISON STUDY
MODEL–MODEL INTERCOMPARISON SCENARIO

II–1. BACKGROUND

The source term for this scenario is hypothetical, but the forest data is based on data for a real forest. The data for this scenario were collated by N Sancharova, Russian Institute of Agricultural Radiology, Obninsk, Russian Federation.

II–2. SOURCE TERM

Spike release and deposition of ^{137}Cs as dry aerosol. Total initial deposition at the top of the canopy is 50 kBq m^{-2}.

Deposition date: 1st May.

II–3. TOPOGRAPHY AND CLIMATE

Deposition is to a uniform area of forest on level ground. The average annual temperature is 5.3°C. January is the coldest month (–8.5°C), and July the warmest (+19.4°C) (Table II-1). The period of average daily temperatures above +10 °C is 140–150 days. The maximum snow cover occurs from the second week of February to the first week of March and reaches 30 cm. The snow cover melts in late March, early April. The annual precipitation varies from 550 to 790 mm. About 70–75% of the total annual precipitation falls during the warm period, from April to October. Small amounts of precipitation of up to 1 mm day^{-1} contribute about 40% of all precipitation during a year.

TABLE II-1. AVERAGE CLIMATIC CHARACTERISTICS OF THE AREA

Month	Rainfall, mm	Temperature, °C
January	25	– 8.5
February	26	– 6.8
March	38	– 3.7
April	44	+4.2
May	51	+14.5
June	70	+17.9
July	79	+19.4
August	71	+18.1
September	53	+12.9
October	50	+3.8
November	40	–1.9
December	33	– 6.3
Annual	580	+5.3

II–4. SOIL CHARACTERISTICS

The main type of soil is soddy-podzolic loamy sand formed from fluvio-glacial sand accumulation. The soils belong to the automorphic group and have a density of 1.2 g cm^{-3}. The main soil mineral is quartz and its content varies from 80% to 95% (in the 0.05–0.01 mm fraction). The clay content is between 0.5% and 1%. More than 95.3% of the soil consists of particles exceeding 0.01 mm (physical sand). The soils of the area are characterized by low natural fertility and unfavourable hydrophysical properties: high water permeability and low water-holding capacity, this causing rapid deep infiltration of melted snow, while considerable quantities of water are evaporated from the upper layers.

TABLE II-2. LITTER AND SOIL CHARACTERISTICS

Horizon	Depth cm	Soil bulk density g cm^{-3}	pH$_{H2O}$	pH$_{KCl}$	Cation exchange capacity Meq 100 g^{-1}	Organic matter %
A_OL *	0–2	0.12	4.5	3.7	76.2	42.58
A_OF *	2–3	0.12	4.9	3.6	74.9	34.03
A_OH *	3–3.5	0.16	5.2	4.0	27.3	22.34
A_OA_1	3.5–5	0.47	5.3	3.9	7.2	2.1
A_1	5–16	0.92	5.2	4.1	4.6	0.78
A_1A_2	16–24	1.16	5.1	4.2	3.7	0.80
B	24–40	1.72	4.9	4.4	2.7	0.05

* Forest litter, mor-moder type.

II–5. TREE CHARACTERISTICS

The dominant species is pine (*pinus sylvestris*) with sparse examples of birch. The rising generation includes pine (*pinus sylvestris*) and birch (*betula pendula*). The average age of the pine trees is 50 years. The birch trees are 40–50 years old. The average age of the trees at the time of contamination is 50 years. The average height of the trees is 20–25 m. The average density of wood biomass is between 120 and 160 metric tonnes per hectare. The growth rates of pine and birch trees, expressed as yearly increases of the diameter and the height, are presented in Table II-3.

TABLE II-3. GROWTH RATES OF AN AVERAGE INDIVIDUAL TREE

Age of trees, years	Height, m/y	Diameter, cm/y	Mass of wood, kg
20	0.34	0.36	12.6
30	0.34	0.34	40.3
40	0.3	0.33	89.5
50	0.25	0.31	161.8
60	0.22	0.28	250.6
70	0.17	0.26	352.4
80	0.15	0.24	475.2
90	0.13	0.22	600.3
100	0.1	0.2	731.4
110	0.08	0.17	852.1
120	0.06	0.13	959.7
130	0.05	0.1	1053.7

TABLE II–4. DYNAMICS OF WOOD MASS FOR AN AVERAGE INDIVIDUAL TREE AND PER UNIT AREA

Age of trees, years	Individual trees (kg)	Per unit area (metric tons ha^{-1})
20	10.8	43
30	34.4	75
40	76.5	119
50	138.3	165
60	214.1	204
70	301.1	235
80	406.1	235
90	513.0	257
100	625.0	275
110	728.2	290
120	820.1	303
130	900.4	317

II–6. UNDERSTOREY CHARACTERISTICS

The total biomass of understorey is about 1.0 kg m^{-2} (10 t ha^{-1}) d.w., including small trees of the rising generation. Shrubs include rowan-tree (*Sorbus aucuparia*), alder black (*Alnus nigra*), buckthorn alder (*Frangula alnus*). The prevailing species of dwarf-shrubs are red raspberry (*Rubus idaeus*) and blackberry (*Rubus trivialis*). The main species of mushrooms are *Boletus edulis*, *Leccinum scabrum*, *Cantharellus cibarius* and *Russula* species.

Grasses are rather sparse. The prevailing species are *Pteridium aquilinum* (fern), *Pyrola rotundifolia*, *Equisetum pratense*, *Calamagrostis epigeios*, *C. arundinacea*, *Deschampsia caespitosa*, *Melica nutans*, *Chamaenerion angustifolium, Majanthemum bifolium.*

Mosses cover 90% of the area. The prevailing species are true mosses (Bryales).

TABLE II-5. BIOMASS OF BERRIES AND MUSHROOMS

Species	Biomass (kg ha^{-1})	
	Fresh weight	Dry weight
Berries:		
Red raspberry (*Rubus idaeus*)	67	7.4
Bilberry (*Vaccinium myrtillus*)	56	6.2
Wild strawberies (*Fragaria vesca*)	15.6	1.7
Mushrooms:		
Suillus luteus	59.7	5.7
Boletus edulis	28.6	2.7
Russula cyanoxantha	17.1	1.6
Cantharellus cibarius	19.1	1.8

II–7. FOREST GAME

The main game species are moose (0.08 animals per km^2), and roe deer (0.06 animals per km^2).

II–8. ENDPOINTS

Participants were required to make predictions of ^{137}Cs activity concentrations in the following forest materials:

- bole wood, Bq kg^{-1};
- total wood (i.e. trunk plus branches), Bq kg^{-1};
- needles, Bq kg^{-1} (annual average);
- other parts of tree, especially bark, Bq kg^{-1};
- soil profile, including litter, Bq kg^{-1} [separate results should be given for each soil layer if possible, as well as for the total organic layer (AoL+AoF+AoH) and the total mineral layer (A_0A_1+A_1+A_1A_2+B)];
- animals, Bq kg^{-1} (annual average for moose and red deer);
- vegetation, Bq kg^{-1} (mushrooms, berries, shrubs and grass).

Activity concentrations were expressed as Bq kg^{-1} **fresh weight** (except for the soil profile).

Participants did not have to report on all the endpoints, but could choose the endpoints they wanted from the list above. Pine (*Pinus sylvestris*) was the main tree of interest, but participants could report on both pine and birch (*Betula pendula*). Similarly, participants could report generically on mushrooms, berries, shrubs and grasses, or on specific species of these.

Each chosen endpoint was considered as:

- a function of time (1 year intervals from 1 to 20 years after the start date).

Calculations were be based on:

- best estimates of parameter input values.

Some modellers also reported results as 95% confidence intervals as well as best estimates.

ANNEX III
MODEL–DATA INTERCOMPARISON STUDY

III–1. GEOGRAPHICAL LOCATION

Ukraine, Zhitomir region, Luginsky district, near v.Rudnya-Povcha, about 130 km to south-west of Chernobyl (51º09'N and 28º35'E).

III–2. SOURCE TERM

Spike release and deposition of ^{137}Cs as dry aerosol. Initial deposition on 1 May 1986 was 555 kBq m^{-2}

III–3. TOPOGRAPHY AND CLIMATE

The average annual temperature is +7.9°C. February is the coldest month (–2.8°C), and July the warmest (+19.8°C) (Table III-1). The period of average daily temperatures above +10°C is 160 days. The maximum snow cover occurs from the second week of February to the first week of March and reaches 15–20 cm. The snow cover melts from the third week of March to the second week of April. The annual precipitation varies from 550 to 710 mm. About 70–75% of the total annual precipitation falls during the vegetative period, from April to October. Small amounts of precipitation of up to 1 mm/day contribute about 50% of annual precipitation. Heavy showers (more than 3mm per rainfall event) are especially typical for summer months.

TABLE III-1. AVERAGE CLIMATIC CHARACTERISTICS OF THE AREA

Month	Rainfall, mm	Temperature, °C
January	24.2	–1.7
February	45.2	–2.8
March	44.1	1.0
April	62.1	7.7
May	57.9	11.7
June	63.5	17.2
July	82.5	19.8
August	56.3	17.7
September	32.1	13.7
October	56.8	7.7
November	29.3	2.8
December	23.0	0.2
Annual	577.0	+7.9

III–4. SOIL CHARACTERISTICS

The main soil type is a soddy podzol developed in sandy loam deposits of fluvio-glacial origin. The average annual depth of groundwater level about 130 cm. The soil is formed from fluvio-glacial sandy-loam deposits. It belongs to the automorphic group and has an average bulk density per 10-cm stratum of 1.15–1.25 g cm^{-3}. The main soil mineral is quartz and its content varies from 60 to 85% (in the 0.05–0.01 mm fraction). The clay content is 0.5%. The soil of the experimental plot No 15 (61) is characterized by low natural fertility and unfavorable hydrophysical properties: high water permeability and low water-holding capacity, thus causing rapid deep infiltration of melted snow, while considerable quantities of water are evaporated from the upper layers. This soddy-podzolic soil is favourable for the growth of boreal tree and plant species – *Pinus sylvestris*, *Vaccinium myrtillus* etc.

Description of soil profile: forest litter thickness reaches 10–15 cm, humus is rough, moder-type. Forest litter mainly consists of needles of *Pinus sylvestris* and residues of mosses (Bryales). Ah-horizon is grey-black, sandy-loam, its thickness is 8–10 cm, with large amounts of roots of dwarf-shrubs, grasses, and rhizoids of mosses. E-horizon (eluvial) is light grey, sandy; its thickness varies from 5 to 10 cm without any roots of plants. Bh-horizon (illuvial) is ferrugineo-brown, loam, dense, with middle diameter roots of *Pinus sylvestris*. Its thickness varies from 5 to 10 cm. The Bi-horizon is yellow brown, sandy-loam or sandy, and exceeds the depth of the ground water. It contains separate big roots of *Pinus sylvestris*.

TABLE III-2. LITTER AND SOIL CHARACTERISTICS

Horizon	Depth, cm	Soil bulk density, g/cm^3	pH $_{H2O}$	pH $_{KCl}$	Cation exchange capacity meq/100g	Organic matter, %
Ol*	0–2	0.026	5,7	5,2	63,5	83,2
Of*	2–6	0.050	5,5	5,0	48,8	75,4
Oh*	6–8	0.093	5,4	4,7	42,3	59,1
Ah	8–18	1.10	5.2	4.6	3,7	4,2
E	18–26	1.30	5.4	4.2	1,3	0,9
Bh (i)	26–36	1.61	5.6	4.7	1,7	1,8
Bi	36–130	1.53	5.7	4.5	0,9	0,1

* Forest litter, humus-moder-type.

TABLE III-3. BULK DENSITY OF SEPARATE STRATUM OF FOREST LITTER AND 2 cm STRATUMS OF MINERAL PART OF THE SOIL

Soil horizon	Depth, cm	Bulk density, g/cm^3
Ah	0–2	0.683
	2–4	0.975
	4–6	0.982
	6–8	1.220
	8–10	1.174
E	10–12	1.340
	12–14	1.340
	14–16	1.346
	16–18	1.345
Bh (i)	18–20	1.450

III–5. TREE CHARACTERISTICS

The dominant species is pine (*Pinus sylvestris*) with sparse birch (*Betula pubescens*). The rising generation includes the same species of trees. The average age of the pine trees is 50 years. The birch trees are about 25–30 years old. The total amount of pine is 1180 trees, the average height of the pine trees is 22 m, diameter 20 cm. The average density of wood biomass is 297.2 metric tonnes per hectare (in fresh weight). The growth rates of pine trees, expressed as yearly increases of the height and the diameter, are presented in Tables III-4 and III-5.

TABLE III-4. GROWTH RATES OF AN AVERAGE INDIVIDUAL TREE

Age of trees, years	Height, m/y	Diameter, cm/y	Mass of wood (trunk with branches), kg (fresh weight)
20	0.42	0.41	26.6
30	0.45	0.40	82.5
40	0.37	0.37	183.8
50	0.30	0.35	297.2
60	0.24	0.31	432.8
70	0.19	0.28	584.8
80	0.16	0.26	747.8
90	0.15	0.24	1006.9
100	0.12	0.22	1341.2
110	0.11	0.21	1565.3
120	0.10	0.18	1759.4
130	0.07	0.14	1943.6

TABLE III–5. DYNAMICS OF WOOD MASS FOR AN AVERAGE INDIVIDUAL TREE AND PER UNIT AREA

Age of trees, years	Individual trees (wood with bark) kg (fresh weight)	Per unit area (metric tonns/ha) (fresh weight)
20	21.6	107
30	71.4	206
40	156.4	297
50	269.9	370
60	397.4	422
70	541.0	467
80	695.6	504
90	941.0	550
100	1258.2	591
110	1472.5	627
120	1658.2	650
130	1833.6	675

III–6. UNDERSTOREY CHARACTERISTICS

Vegetation belongs to floristic association Molinio-Pinetum J.Mat. 1981, Union Dicrano-Pinion Libl. 1933, Ordo Vaccinio-Piceetalia Br.-Bl. 1939 em K.Lund 1967, Class Vaccinio-Piceetea Br.-Bl. 1939. This association is wide spread in Central Polessie of Ukraine (about 40% of total forest cover). The understorey vegetation layer is dense, with a projective cover of 70–75%, representing Vaccinium myrtillus (60–65%), Vaccinium vitis-idaea (5–10%), Vaccinium uliginosum (1–3%), Molinia caerulea (1–3%), Melampyrum pratense (1%), Dryopteris carthusiana (1%), separate plants – Ledum palustre, Equisetum sylvaticum, Luzula pilosa, Lysimachia vulgaris, Calluna vulgaris, Potentilla erecta et al. Moss cover is dense (with projective cover 90–98%), consisting of Pleurozium schreberi (60–65%), Dicranum polysetum (30–33%), Polytrichum commune (1–5%).

The total biomass of understorey vegetation is about 1,3 kg m^{-2} (13,0 t/ha, d.w.), including small trees of the rising generation, and shrubs of Sorbus aucuparia and Frangula alnus (about 0.1 kg m^{-2}) d.w. Biomass of mosses is 1,0 kgm^{-2}, dwarf-shrubs (mainly Vaccinium myrtillus – 0.12–0.14 kg m^{-2}). The main species of mushrooms are Xerocomus badius, Cantharellus cibarius, Russula paludosa, Suillus luteus, Boletus edulis.

TABLE III–6. BIOMASS OF BERRIES AND MUSHROOMS

Species	Biomass (kg/ha) Aboveground phytomass, (d.w.)	Berries (f.w.)	Berries (d.w.)	Mushrooms, fruitbodies (d.w.)
Berry species				
Vaccinium myrtillus	1200	320	31,8	
V.vitis-idaea	20	–	–	
V.uliginosum	2	–	–	
Mushrooms				
Suillus luteus	–	–	–	0.1
Cantharellus cibarius	–	–	–	0.2
Russula paludosa	–	–	–	1.3
Xerocomus badius	–	–	–	2.0
Boletus edulis	–	–	–	0.1

III–7. FOREST GAME

The main game species is roe deer (0.03 animals per km^2).

III–8. ENDPOINTS

Participants were required to make predictions of ^{137}Cs activity concentrations in the following forest materials:

- Wood, Bq/kg dry weight;
- Annual shoots, Bq/kg dry weight;
- Annual Needles, Bq/kg dry weight;
- Total bark[*] , Bq/kg dry weight;
- Soil profile, Bq/kg dry weight;
- Roe deer , Bq/kg in muscles (fresh weight), September–October;
- Mushrooms, Bq/kg dry weight;
- Bilberries, Bq/kg dry weight.

Participants did not have to report on all the endpoints, but could choose the endpoints they wanted from the list above. Participants could report generically on mushrooms and berries, or on specific species of these.

Each chosen endpoint was considered as a function of time (1 year intervals from 1986 to 1998). Calculations should have been based on best estimates of parameter input values. Results should have been reported as either as 95% confidence intervals, or as best estimates.

---

[*] Total bark is defined as the internal plus external bark. Internal bark is the cambium; *i.e.* the physiologically active part between the external bark and the wood.

TABLE III-7. EXPERIMENTAL DATA FOR MODEL–DATA INTERCOMPARISON STUDY

(Provided by A Orlov, Polesskaya Forest Scientific Research Station, Zhitomir, Ukraine)

Specific activity of ^{137}Cs in samples on experimental plot ¹ 15 (61)							
Type of sample	Specific activity of ^{137}Cs in samples, Bq kg^{-1}						
	1991	1992	1993	1994	1995	1996	1997
Pine-tree (d.w.)							
Bark external (d.w.)	6667	5080	4030	3977	3511	3090	3500
	5974	5163	5204	3973	4901	3960	3800
	5488	6740	3975	4950	4443	4140	2750
Mean	6043	5661	4403	4300	4285	3730	3350
Bark total (d.w.)	6510	4500	5170	4700	4490	3700	4690
	6030	6360	4470	3720	347	4600	3180
	5910	6750	5990	5380	539	4660	4640
Mean	6150	5870	5210	4600	1792	4320	4170
Bark internal (bast) (d.w.)	5670	7700	7295	6890	6754	6946	11694
	7140	5810	6230	6780	9315	10553	7701
	4950	8090	7343	6130	6485	9642	10176
Mean	5920	7200	6956	6600	7518	9047	9857
Wood (without bark)	1284	1424	1394	1280	1409	2726	2300
	2017	948	1561	1409	1663	2529	2252
	1694	1369	1191	1670	1407	3178	3068
Mean	1665	1247	1382	1453	1493	2811	2540
Annual shoots (d.w.)	21776	38631	10316	17903	17611	54681	47380
	23708	31692	10689	12956	11873	63621	63510
	15935	37311	12739	19757	15069	45555	76610
Mean	20473	35878	11248	16872	14851	54619	62500
Annual needles (d.w)	19250	27951	11554	17816	13062	38299	50610
	20280	42029	15340	13277	20042	47282	41010
	23740	36172	15397	15737	16522	38688	38370
Mean	21090	35384	14097	15610	16542	41423	43330
Berries (f.w.)							
Vaccinium myrtillus	7450	6380	5813	4960	4684	7899	7420
	7480	5760	6477	9871	7469	5303	3740
	14530	3500	6551	5591	9676	3740	6760
	11800	6760	8296	9502	5333	8257	3920
	8990	9050	7833	7261	7163	4744	5960
Mean	10050	6290	6994	7437	6865	5988.6	5560
All abogr phytom (d.w)	32905	48930	42200	34800	37120	42755	25250
	60055	57036	67300	44860	51540	40145	26369
	59755	68779	35100	23800	51710	31040	40604
	56070	51359	32600	33070	42510	34340	51883
	80852	57499	63700	35260	29410	42526	56670
Mean	57927	56720	48180	34358	42458	38161.2	40155.2
Mushrooms							
Xeroc badius (d.w.)	3189000	6530000	6141000	3777000	5260000	5910000	5660000
	4337400	2950000	6231000	5100000	1170000	4800000	4450000
	3755100	2310000	1571500	2130000	1700000	1160000	1120000
	7112800	2580000	4334200	2095000	5650000	3970000	4640000
	1932600	6040000	25050	6062000	3830000	1940000	1520000
Mean	4065380	4082000	3660550	3832800	3522000	3556000	3478000

TABLE III–7. (cont.)

Specific activity of ^{137}Cs in samples on experimental plot ¹ 15 (61)

Type of sample	Specific activity of ^{137}Cs in samples, Bq kg^{-1}						
	1991	1992	1993	1994	1995	1996	1997
Suillus luteus (d.w.)	4583500		1870000	3060000		4059000	3303000
	3553800		3790000	4910000		3399000	2148000
	3032600		4360000	2560000		2676000	4785000
Mean	3723300		3340000	3510000		3378000	3412000
Canth cabarius (d.w.)	1294500		1411000		109000	555300	
	912700		984000		89000	1279800	
	1304700		1130000		144000	878200	
	1400000		1126000		97000	1016200	
	1266000		794000		110000	887500	
Mean	1235580		1089000		109800	923400	
Boletus edulis (d.w.)	895300			809300			861200
	317200			457400			518700
	672100			483200			513700
Mean	628200			583300			631200
Russ paludosa (d.w.)	1705600	1624200	1758400	1341500	959300	2420000	1516000
	762700	961200	372100	584800	2626900	1236000	1531000
	1849600	2159500	2083000	2835100	1397600	655000	2164000
	1664800	1193700	931900	2589300	1930700	1816000	747000
	3284800	477500	2038600	1278300	1229000	1592000	372000
Mean	1853500	1283220	1436800	1725800	1628700	1543800	1266000
Roe-deer (muscles)	63550	62330	–	60355	–	55330	56280
Soil profile (d.w.)							
Ol	9985	11141	10835	10599	6780	7007	5797
	11871	6324	7975	6278	9310	9820	7291
	7649	10375	8295	7393	7400	5676	9532
Mean	9835	9280	9035	8090	7830	7501	7540
Of	87352	128201	81215	84221	61300	51675	58080
	125199	74984	96726	116016	71360	48783	50360
	88580	94514	116434	85153	59640	40704	51160
Mean	100377	99233	98125	95130	64100	47054	53200
Oh	59906	66690	66620	46485	65090	49055	111840
	48263	36480	48370	53060	75150	64584	78720
	47246	55380	45660	66685	42730	73156	81540
Mean	51805	52850	53550	55410	60990	62265	90700
Ah 0–2 cm	7733	4620	5230	7218	8030	5842	6180
	5513	6422	7910	6254	5020	8310	4660
	4349	7948	4650	5620	8100	10031	8030
Mean	5865	6330	5930	6364	7050	8061	6290
2–4 cm	2402	1794	2300	2026	1790	1907	3790
	1432	1869	2294	1781	2950	2216	2460
	1926	2592	1310	2586	2070	3098	2210
Mean	1920	2085	1968	2131	2270	2407	2820
4–6 cm	374	615	522	516	644	1204	1202
	339	365	392	287	726	793	1778
	562	619	346	520	535	718	1370
Mean	425	533	420	441	635	905	1450

TABLE III–7. (cont.)

Specific activity of ^{137}Cs in samples on experimental plot ¹ 15 (61)							
Type of sample	Specific activity of ^{137}Cs in samples, Bq kg^{-1}						
	1991	1992	1993	1994	1995	1996	1997
6–8 cm	224	225	291	249	369	766	1037
	192	210	168	236	510	526	962
	133	315	231	268	351	457	836
Mean	183	250	230	251	410	583	945
8–10 cm	155	135	191	138	250	258	358
	141	165	170	211	222	382	360
	178	222	116	152	188	377	398
Mean	158	174	159	167	220	339	372
10–12 cm	140	131	137	69	161	289	209
	195	146	169	73	121	304	216
	106	92	93	110	177	166	223
Mean	147	123	133	84	153	253	216
12–14 cm	167	140	105	199	156	234	196
	125	195	163	127	148	206	190
	158	145	206	175	236	181	109
Mean	150	160	158	167	180	207	165
14–16 cm	45	34	37	38	139	201	95.9
	25	42	52	36	114	283	75.9
	35	53	61	58	77	182	121
Mean	35	43	50	44	110	222	97.6
16–18 cm	42	39	51	47	154	248	51.3
	36	36	48	35	109	235	42.5
	42	24	51	56	97	231	60.4
Mean	40	33	50	46	120	238	51.4
18–20 cm	28	23	51	15	52	282	210
	22	22	42	13	91	158	345
	19	15	27	20	97	232	399
Mean	23	20	40	16	80	224	318

ANNEX IV
SECOND MODEL–MODEL INTERCOMPARISON STUDY (SCENARIO 3)

IV–1. SOURCE TERM

The source term is a continuous release of ^{137}Cs from loosely tipped radioactive waste disposed in a number of trenches that are now covered by soil caps. Trees grow on the site, with the roots penetrating through the soil caps directly into the waste. We would like to know the activity concentration of ^{137}Cs in various parts of the trees up to 200 years after they have regenerated naturally on the trench caps.

It is assumed that the radioactive waste is disposed in a minimally engineered facility consisting of 10 trenches (two rows of five units) orientated perpendicular to the groundwater flow (see Figure IV-1). The trenches are adapted from a previous IAEA study on quantitative acceptance criteria for near surface disposal of radioactive waste [1]. Relevant parameters for the waste and the trenches are as follows (also see Figure IV-1):

10 trenches (two rows of five units)

Dimensions of trench:
  internal length = 100 m
  internal width = 15 m
  internal depth = 6 m
  distance between two trenches (edge to edge) = 20 m.

Total area of the site = 195 × 260 m (i.e. 5.07E+4 m^2)

Homogeneous waste backfilled with native, sandy soil:
  total porosity = 0.5
  waterfilled porosity = 0.4
  hydraulic conductivity = 10^{-5} $m.s^{-1}$
  total bulk density = 500 $kg.m^{-3}$
  activity concentration of ^{137}Cs in waste = 1 kBq kg^{-1} dry weight.

Radioactive decay of ^{137}Cs can be assumed in calculations. It should also be assumed that the potential bioavailability of ^{137}Cs within the waste material is the same as that in the mineral soil layer.

Cap made of natural material (i.e. made of compacted, sandy soil):
  thickness = 1 m
  kinematic and total porosity = 0.3
  hydraulic conductivity = 10^{-7} $m.s^{-1}$
  bulk density = 1500 $kg.m^{-3}$

Soil properties:
  unsaturated zone thickness (temperate) = 2 m below base of disposal unit
  average moisture content of the unsaturated zone = 0.15
  kinematic and total porosity = 0.3
  bulk density = 1700 $kg.m^{-3}$
  hydraulic conductivity = 10^{-5} $m.s^{-1}$
  hydraulic gradient = 1 in 50
  thickness of the saturated zone = 15 m
  longitudinal dispersivity = distance to outlet / 10 (m)
  transverse dispersivity = distance to outlet / 50 (m).
  water table: 2 m below the bottom of the cap that covers the waste

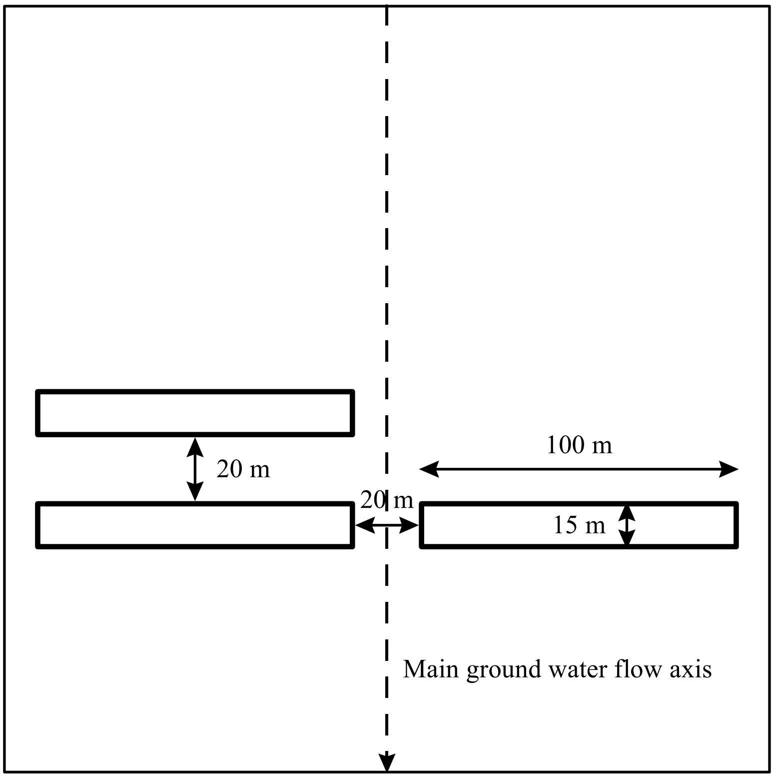


FIG. IV-1. Trench disposal facility.

IV–2. CLIMATE

temperate precipitation = 1000 mm.y^{-1} (yearly averaged)
temperate actual evapotranspiration = 400 mm.y^{-1}

Closure of facility May

IV–3. TREE CHARACTERISTICS

There are no trees at time zero (i.e. when the trench cap is put in place), but there is natural regeneration of vegetation, including tree saplings, within the first year after closure. The dominant tree species is pine (*Pinus sylvestris*) with sparse examples of birch. Two hundred years after closure the average height of the trees is 20–25 m. The average density of wood biomass on the trenches is between 10 and 20 kg m^{-2}. No information on tree growth rates is available over this period.

The roots of the trees penetrate through the soil caps directly into the waste. It should be assumed that the potential bioavailability of ^{137}Cs within the waste material is the same as that in the mineral soil layer. The distribution of the roots with depth at various tree ages is given in Tables IV-1 and IV-2. The following general observations on the distribution of tree roots with depth should also be kept in mind:

- pine root growth rate decreases with age;
- pine roots reach half a maximum depth after 10–15 years;
- roots distribution of trees older than 40–60 years does not change significantly with time.

TABLE IV-1. ROOTS DISTRIBUTION FOR 12 YEARS OLD PINE

Horizon	Layer, cm	Roots dry weight, g/tree large roots (d>1mm)	Small roots (d<1 mm)
A_1	1–11	140,7	64,3
A–B	11–21	230,2	38,0
	21–31	48,5	22,4
B_1	31–41	27,3	14,1
	41–61	37,5	21,9
B_2	61–81	24,6	10,7
	81–101	3,6	1,2
	101–121	8,7	0,5
C	121–151	11,4	3,4
	151–171	5,8	1,8
	171–201	6,5	2,0
A,B,C	1–201	545	180

TABLE IV-2. ROOTS DISTRIBUTION FOR 33 YEARS OLD PINE

Horizon	Layer, cm	Roots dry weight, g/tree large roots (d>1mm)	Small roots (d<1 mm)
A	2–12	290,7	92,2
A–B	12–22	607,3	37,8
	22–32	406,9	29,6
	32–42	298,9	21,6
B	42–62	236,9	20,7
	62–82	146,9	20,9
	82–102	92,9	5,8
	102–122	55,1	4,2
	122–152	87,4	6,6
	152–172	72,7	1,6
C	172–202	117,8	3,8
	202–222	34,0	1,9
	222–252	50,9	2,8
A,B,C	1–252	2498	250

IV–4. UNDERSTOREY CHARACTERISTICS

The total biomass of understorey is about 1.0 kg m^{-2} (10 t/ha) d.w. The prevailing species of dwarf-shrub is bilberry (*Vaccinium myrtillus;* 0.12–0.14 kg m^{-2}). The main species of mushrooms are *Boletus edulis* and *Xerocomus badius.*

TABLE IV–3. BIOMASS OF BERRIES AND MUSHROOMS

Species	Biomass (kg/ha) Aboveground phytomass, (d.w.)	Berries (f.w.)	Berries (d.w.)	Mushrooms, fruitbodies (d.w.)
Berry species				
Vaccinium myrtillus	1200	320	31,8	
Mushrooms				
Xerocomus badius	–	–	–	2.0
Boletus edulis	–	–	–	0.1

IV–5. ENDPOINTS

Participants are required to make predictions of ^{137}Cs activity concentrations in the following forest materials:

- Total tree, Bq/kg;
- Total wood (i.e. trunk plus branches), Bq/kg;
- Needles, Bq/kg (annual average);
- Total bark, Bq/kg;
- Cap Bq/kg (for 1m depth at 10 cm intervals from the surface, if possible);
- vegetation, Bq/kg (mushrooms and berries).

Activity concentrations should have been expressed as Bq/kg **fresh weight** (except for the cap, which should have been specified in Bq/kg dry weight).

For each endpoint, results were reported at 1 year intervals for years 0–20 after closure, thereafter at 10 year intervals for years 30–200.

As a minimum, final activity concentrations 200 years after closure should have been given.

Calculations should have been based on best estimates of parameter input values. If possible, results should have been reported as:

- 95% confidence intervals; or
- best estimates.

References

[1] Derivation of quantitative acceptance criteria for disposal of radioactive waste to near surface facilities: development and implementation of an approach, IAEA draft report version 3.0, March 1999.

[2] KALININ, M.I., Formation of tree root system, Moscow, p152 (1983).

[3] RAKHTEENKO, I.N., Root systems of trees and bushes, Moscow-Leningrad, p108 (1952).

MEMBERS OF THE FOREST WORKING GROUP

The following persons attended the meetings of the Forest Working Group (October 1998 to May 2000):

G. Ågren
Swedish University of Agricultural Sciences
Department of Ecology and Env Research
Box 7072, SE-750 07 Uppsala, SWEDEN
Tel: +46 18 67 2449
Fax: +46 18 67 3430
Email: goran.agren@eom.slu.se

P. Armand
CEA/DAM/DIF/DASE/SRCE
Bâtiment G
B.P. 12, F-91680 Bruyeres-le-Chatel, FRANCE
Tel: +33 (1) 6926-4536
Fax: +33 (1) 6926-7065
Email: armand@ldg.bruyeres.cea.fr

R. Avila
Swedish Radiation Protection Institute
S-171 16 Stockholm
SWEDEN
Tel: +46 8 729 7211
Fax: +46 8 729 7108
Email: rodolfo.avila@ssi.se

M. Belli
Environmental Protection Agency
Agenzia Nazionale per la Protezione dell'Ambiente (ANPA)
Via Vitaliano Brancati 48
I-00144 Rome, ITALY
Tel: +390 (6) 5007-2952/2869/2924
Fax: +390 (6) 5007-2313/2856
Email: belli@anpa.it

R. Bergman
National Defence Research Establishment
Department of NBC-Defence
S-901 82 Umea
SWEDEN
Tel: +46 90 106 600
Fax: +46 90 106 803
Email: bergman@ume.foa.se

G. Brachet
CEA/DAM/DASE
B.P. 12
F-91680 Bruyeres-Le-Chatel
FRANCE
Tel: +33 (1) 6926-5029
Fax: +33 (1) 6926-7023
Email: gbrachet@dase.bruyeres.cea.fr

A. Bulgakov
SPA "Typhoon"
Institute of Experimental Meteorology
82 Lenin Avenue
Kaluga Region, 249020 Obninsk
RUSSIAN FEDERATION
Tel: +7 08439 71914 or +7 08439 40910
Fax: +7 08439 71896 or +7 095 255 2225
Email: konoplev@iem.obninsk.ru

F. Carini
Faculty of Agricultural Sciences
Institute of Agricultural & Env Chemistry
Università Cattolica del Sacro Cuore
Via Emilia Parmense 84, I-29100 Piacenza
ITALY
Tel: +390 (523) 599-156
Fax: +390 (523) 599-358
Email: fcarini@pc.unicatt.it

P. Calmon
IRSN/DPRE/SERE/LMODE
Institut de Radioprotection et de Sûreté
Nucléaire (IRSN), Centre de Cadarache
B.P. 1, F-13108 Saint-Paul-lez-Durance, Cedex
FRANCE
Tel: +33 (442) 257-135
Fax: +33 (442) 256-292
Email: philippe.calmon@ipsn.fr

J. Droppo
Battelle Pacific Northwest National Laboratory
Battelle Boulevard,
P.O. Box 999
WA 99352 Richland
UNITED STATES OF AMERICA
Tel: +1 (509) 376-7652
Fax: +1 (509) 373-0335
Email: james.droppo@pnl.gov

A. Dvornik
Forest Institute of Belorussian Academy
of Science
71 Proletarskaya Street, 246654 Gomel
BELARUS
Tel: +375 232 53 8341
Fax: +375 232 53 5389
Email: dvornik@fi.gomel.by

K. Eged
Department of Radiochemistry
University of Veszprém
P.O. Box 158 H-8201, H-8200 Veszprém
HUNGARY
Tel: +36 (88) 427-681
Fax: +36 (88) 427-681
Email: egedk@fiz5.mars.vein.hu

S. Fesenko
Russian Institute of Agricultural Radiology
& Agroecology
Kievskoe shosse 1
Kaluga Region, 249020 Obninsk
RUSSIAN FEDERATION
Tel: +7 (08439) 67205/64802
Fax: +7 (095) 255-2225
Email: acr@meteo.ru

M. Frissel
Private Consultant
Torenlaan 3, NL-6866-BS Heelsum
THE NETHERLANDS
Tel: +31 (317) 312-248
Fax: +31 (317) 312-248
Email: frisselm@bart.nl

F. Gera (Retired) (formerly of the IAEA)
Via Monte dell'Ara 14
0060 Formello, Rome
ITALY
Tel: +390 (6) 3735-2582
Fax: +390 (6) 3735-2582
Email: F_Gera@yahoo.com

S. Geuens
Radiation Protection Research Unit
Belgium Nuclear Research Center (SCK/CEN)
Boeretang 200
B-2400 Mol
BELGIUM
Email: sgeuens@sckcen.be

A. Golubev
RFNC-VNIIEF
Mira pr., 37
Nizhni Novgorod Region, 607190 Sarov
RUSSIAN FEDERATION
Tel: +7 (831) 304-0995
Fax: +7 (831) 305-3808
Email: avg@dc.vniief.ru

F. Goor
Belgium Nuclear Research Center (SCK/CEN)
Boeretang 200
B-2400 Mol
BELGIUM
Tel: +32 (14) 335-282
Fax: +32 (14) 580-523
Email: fgoor@sckcen.be

U. Kautsky
Swedish Nuclear Fuel and Waste
Management Company
P.O. Box 5864,
S-102 48 Stockholm
SWEDEN
Tel: +46 8 459 8419
Fax: +46 8 661 5719
Email: skbuk@skb.se

Z. Kis
Department of Nuclear Medicine
County Hospital Veszprém
Kórház út 1, H-8201 Veszprém
HUNGARY
Tel: +36 (88) 420-211
Fax: +36 (88) 421-457
Email: kissz@fiz5.mars.vein.hu

A. Konoplev
Institute of Experimental Meteorology
SPA "Typhoon"
82 av Street
Kaluga Region, 249020 Obninsk
RUSSIAN FEDERATION
Tel: +7 08439 71896
Fax: +7 08439 71896
Email: konoplev@hotmail.com

V. Krasnov
Polesskya Agro-forest-ameliorative Scientific
Research Station
UA-262004 Zhitomir
UKRAINE
Tel: +38 412 268 638
Fax: +38 412 268 628
Email: station@impuls.zhitomir.ua

I. Linkov
Menzie-Cura & Associates, Inc.
1 Courthouse Lane, Suite 2
MA 01824-1734 Chelmsford
UNITED STATES OF AMERICA
Tel: +1 (978) 453-4300 x15
Fax: +1 (978) 453-7260
Email: ilinkov@ma.ultranet.com

L. Moberg
Swedish Radiation Protection Institute
S-171 16 Stockholm
SWEDEN
Tel: +46 8 729 7100
Fax: +46 8 729 7108
Email: leif.moberg@ssi.se

M.B. Oncsik-Biro
Irrigation Research Institute
Szabadsagu u. 2, H-5540 Szarvas
HUNGARY
Tel: +36 (66) 311-574
Fax: +36 (66) 311-178
Email: oncsik@oki.ince.hu

A. Orlov
Polesskya Agro-forest-ameliorative
Scientific Research Station, 262004 Zhitomir
UKRAINE
Tel: +380 412 268 638
Fax: +380 412 268 628
Email: station@impuls.zhitomir.ua

A. Rantavaara
Finnish Centre for Radiation & Nuclear Safety
Research Department, Foodchain Laboratory
P.O. Box 14 (Laippatie 4)
FIN-00881 Helsinki
FINLAND
Tel: +35 89 7598 8436
Fax: +35 89 7598 8498
Email: aino.rantavaara@stuk.fi

T. Riesen
Division for Radiation Protection & Waste Management,
OSUA/105
Paul Scherrer Institute (PSI)
CH-5232 Villigen PSI
SWITZERLAND
Tel: +41 (56) 310-2341
Fax: +41 (56) 310-2309
Email: thomas.riesen@psi.ch

N. Sanjarova
Russian Institute of Agricultural Radiology
& Agroecology
RU 249020 Obninsk
RUSSIAN FEDERATION
Tel: +7 084 39 67 205
Fax: +7 095 255 22 25
Email: acr@wdc.meteo.ru

T. Sazykina
Institute of Experimental Meteorology
SPA "Typhoon", 82 Lenin Street
Kaluga Region, 249020 Obninsk
RUSSIAN FEDERATION
Tel: +7 (08439) 71698/37536
Fax: +7 (08439) 40910
Email: ecomod@obninsk.com

G. Shaw
Imperial College of Science Technology
& Medicine
Centre for Analytical Res. in the Environment
Silwood Park, Sunningdale
Ascot, Berkshire SL5 7TE
UNITED KINGDOM
Tel: +44 1344 294 277
Fax: +44 1344 249 31
Email: gg.shaw@ic.ac.uk

F. Siclet
Electricité de France
Département Environement (EDF DER)
6, Quai Watier
B.P. 49
F-78401 Chatou Cédex
FRANCE
Tel: +33 (1) 3087-7847
Fax: +33 (1) 3087-7336
Email: francoise.siclet@edf.fr

S. Spiridonov
Russian Institute of Agricultural Radiology &
Agroecology
Kaluga Region
249020 Obninsk
RUSSIAN FEDERATION
Tel: +7 084 39 248 02
Fax: +7 095 255 2225
Email: acr@wdc.meteo.ru

M. Steiner
Institute for Radiation Hygiene,
Federal Office for Radiation Protection
Ingolstädter Landstrasse 1
Oberschleissheim, D-85764 Neuherberg
GERMANY
Tel: +49 89 3160 3369
Fax: +49 89 3160 3111
Email:msteiner@bfs.de

E. Steinnes
Norwegian University of Science & Technology
Department of Chemistry
N-7034 Trondheim
NORWAY
Tel: +47 735 96237
Fax: +47 735 96940
Email: eiliv.steinnes@chembio.ntnu.no

F. Strebl
Department für Umweltforschung
Österreichisches Forschungszentrum Seibersdorf GesmbH
A-2444 Seibersdorf
AUSTRIA
Tel: +43 (1) (2254) 780-3579
Fax: +43 (1) (2254) 780-3653
Email: strebl@arcs.ac.at

Y. Thiry
Belgium Nuclear Research Center (SCK/CEN)
Radiation Protection Research Unit
Boeretang 200
B-2400 Mol
BELGIUM
Tel: +32 14 335 282
Fax: +32 14 320 372
Email: ythiry@sckcen.be

H. Tsukada
Department of Radioecology
Institute for Environmental Sciences
1-7 Ienomae, Obuchi
Rokkasho-mura, Kamikita-gun, 039-3212 Aomori
JAPAN
Tel: +81 (175) 711-457
Fax: +81 (175) 711-492
Email: hirot@ies.or.jp

U. Tveten
Institute for Energy Technology (IFE)
P.O. Box 40, N-2007 Kjeller
NORWAY
Tel: +47 (63) 806-000
Fax: +47 (63) 816-356
Email: ulft@ife.no / ulftvet@online.no

E. Uspenskaya
All Russian Res. Institute of Nature Protection
113628 Moscow
RUSSIAN FEDERATION
Tel: +7 095 423 1301
Fax:+7 095 423 2322
Email: rinpro@glasnet.ru

H. Vandenhove
Radiation Protection Research Unit
SCK/CEN
Boeretang 200, B-2400 Mol
BELGIUM
Tel: +32 14 335 282
Fax: +32 14 320 372
Email: hvandenhove@sckcen.be

E. van der Stricht
International Union of Radioecology (IUR)
27 Domaine de Brameschhof
L-8290 Kehlen
LUXEMBOURG
Tel: +352 306 136
Fax: +352 306 137
Email: estricht@pt.lu

A. Venter
Enviros QuantiSci
Building D5 Culham Science Park
Abingdon, Oxfordshire OX14 1BH
UNITED KINGDOM
Tel:+44 (0)870 162 8811 or +44 (0)870 162 8804
Fax:+44 (0)870 162 8828
Email: ansie.venter@enviros.com

J. Wendt
Technical Research Centre
P.O. Box 2106 (Espoo Vuorimiehentie 5)
FIN-02044VTT
FINLAND
Tel: +358 (9) 4561
Fax: +358 (9) 456-7001
Email: joachim.wendt@vtt.fi

S. Yoshida
Environmental & Toxicological Sciences
Research Group
National Institute of Radiological
Sciences (NIRS)
9-1, Anagawa 4-chome
Inage-ku, Chiba-shi, 263-8555 Chiba
JAPAN
Tel: +81 (43) 206-3255
Fax: +81 (43) 206-3267
Email: s_yoshida@nirs.go.jp

G. Zibold
Fachhochschule Ravensburg-Weingarten
Postfach 1261
D-88241 Weingarten
GERMANY
Tel: +49 (751) 501-562
Fax: +49 (751) 49240
Email: zibold@fbp.fh-weingarten.de

CONTRIBUTORS TO DRAFTING AND REVIEW

Shaw, G.	Imperial College of Science Technology and Medicine, United Kingdom
Venter, A.	Enviros QuantiSci, United Kingdom
Avila, R.	Swedish Radiation Protection Institute (SSI), Sweden
Thiry, Y.	Belgium Nuclear Research Center (SCK/CEN), Belgium
Goor, F.	Belgium Nuclear Research Center (SCK/CEN), Belgium
Linkov, I.	Menzie-Cura and Associates, Inc., United States of America
Steiner, M.	Federal Office for Radiation Protection, Germany
Orlov, A.	Polesskya Agro-forest-ameliorative Scientific Research Station, Ukraine
Gera, F.	International Atomic Energy Agency (formerly)

Meetings

BIOMASS Forest WG Planning Meeting, Vienna, Austria: 26–27 March 1998
BIOMASS Plenary 1998 Meeting, Vienna, Austria: 5–9 October 1998
BIOMASS Forest WG Spring 1999 Meeting, Stockholm, Sweden: 24–27 May 1999
BIOMASS Plenary 1999 Meeting, Vienna, Austria: 6–10 October 1999
BIOMASS Forest WG Spring 2000, Brussels, Belgium: 19–21 May 2000
BIOMASS Plenary 2000 Meeting, Vienna, Austria: 7–9 November 2000